WARZYN

№ 0000061

Report
Project No. 13160

Volume 1 of 2
Remedial Investigation
Winnebago Reclamation Landfill
Rockford, Illinois

Prepared for:

Pagel's Pit Landfill
Potentially Responsible Parties
Rockford, Illinois

Prepared by:

Warzyn Inc.
Chicago, Illinois

March 1991

REMEDIAL INVESTIGATION REPORT

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

Prepared by:

WARZYN INC.
Chicago, Illinois

on behalf of:

Pagel's Pit Potentially Responsible Parties
for
U.S. EPA Region V

March 1991

Project No. 13160

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Project Manager

TABLE OF CONTENTS

TABLE OF CONTENTS
(Continued)

TABLE OF CONTENTS
(Continued)

Page

TABLE OF CONTENTS
(Continued)

TABLE OF CONTENTS
(Continued)

Page

LIST OF TABLES

TABLE OF CONTENTS
(Continued)

LIST OF FIGURES

TABLE OF CONTENTS
(Continued)

LIST OF DRAWINGS

LIST OF APPENDICES

EXECUTIVE SUMMARY

This document has been prepared to provide a summary and evaluation of data collected during the Remedial Investigation (RI) performed at the Winnebago Reclamation Landfill (WRL Site or Site), also known as Pagel's Pit Landfill, located near Rockford, Illinois. The RI performed at the WRL Site was accomplished in two phases. At the completion of Phase I, the Interim Groundwater Quality Evaluation (IGQE) was prepared and submitted to the United States Environmental Protection Agency (U.S. EPA). The Phase II Investigation was performed based upon the recommendations in the IGQE and approved by the U.S. EPA.

The WRL Site, located about 5 miles south of Rockford, Illinois, is situated on a topographic high between Killbuck Creek to the west and unnamed intermittent streams to the north and south. The WRL Site is an active solid waste landfill with a leachate collection system and a landfill gas collection system, permitted by the state of Illinois. The WRL Site is located in a rural setting with land use around the Site a mix of industrial, agricultural, commercial, and rural residential. East (upgradient) of the WRL Site is the former Acme Solvent Reclaiming Inc. Site, an industrial waste disposal site which operated from 1960 to 1973 and was used for the disposal of, among other wastes, solvent still-bottoms, non-recoverable solvents, paints, and oils in unlined pits. The Rockford Skeet Club is across Lindenwood Road to the northeast. A septic tank pumping business is located to the west, a private hunt club to the southwest, and a limestone quarry to the northeast of the WRL Site. There are residences scattered within 1/2 mile of the site to the north, south, southwest, and southeast.

The surficial unconsolidated deposits in the WRL Site area are predominantly glacial drift ranging from a thin mantle over the dolomite bedrock at the Acme Solvent Site to greater than 70 feet filling the bedrock valley west of the Site. The unconsolidated deposits are predominantly sand and gravel underneath and north of the WRL Site with a silty clay to the south of the Site. The underlying bedrock surface is highly variable due to paleoerosional features. The dolomite bedrock is generally fractured but the intensity is variable. Chert layers or nodules were commonly noted on boring logs as were vugs (void spaces), but cavernous zones were not reported.

Groundwater flow within the dolomite bedrock is believed to be largely controlled by fracture zones. The variable intensity of fracturing with depth can lead to preferential flow through a more permeable, more fractured zone. The presence of a high permeability zone between the WRL and Acme Solvent Sites is inferred from geologic data, geophysical data, pump test results, permeability test results, water level observations, and groundwater quality data.

Based upon the data collected during the RI and other available information as referenced in this report, it is concluded that:

- Groundwater flows under water table conditions in fractured dolomite bedrock from the uplands east of the WRL Site to the west towards the unconsolidated sediments of the Killbuck Creek Valley. Groundwater discharges to the Creek, but also passes beneath it.

- The WRL leachate is characterized by its high inorganic component, particularly chloride. The WRL leachate generally contained aromatic VOCs such as benzene, toluene, ethylbenzene and xylenes at higher concentrations then chlorinated compounds such as PCE and TCE.

- The WRL leachate groundwater plume is characterized by the presence of elevated chloride and the relative enrichment of the cations sodium and potassium.

- Elevated chloride concentrations in the groundwater at locations around the WRL Site define the plume associated with the landfill leachate. These general areas include:

  - The northwest quadrant of the landfill defined by wells B15R, MW106, P1, P4R, and G116A

  - In the vicinity of wells G110 and G114

  - In the vicinity of well G115

- In the northwest quadrant of the WRL Site, the leachate plume is present only in the shallow groundwater (near well nest B15/B15R/B15P), but has reached the deeper zones downgradient at well nest P1/MW106 east of Killbuck Creek. The plume had migrated west beneath Killbuck Creek in the deeper groundwater zone but the shallow zone appears unaffected. Lower chloride concentrations at well nest P1/MW106 and G116A were observed in sampling Rounds 3 and 4.

- The WRL leachate plume inorganic component tends to contain elevated chlorides, sodium, potassium, magnesium, iron, and manganese. Other constituents sometimes associated with the WRL leachate plume include: total phenolics, cyanide, arsenic, barium, cobalt, copper, lead, nickel, silver, zinc, and vanadium. These constituents were either not detected or present only at low levels in samples from well G116A, indicating these constituents are attenuated in the aquifer as expected.

- The groundwater upgradient of the WRL Site has been impacted by VOCs with the highest concentrations observed in samples from well B4. The WRL leachate plume has overprinted VOC impacted groundwater, since VOCs are found both inside and outside of the WRL leachate plume.

- The VOCs observed in RI samples can be divided into three general groups based upon their detection in groundwater in an east to west direction (i.e., direction of groundwater flow). VOCs detected at well B4, VOCs associated with wells upgradient of the WRL Site near Lindenwood Road, and VOCs present only at low levels in the WRL leachate plume.

- Potential sources of VOCs other than the WRL leachate plume include releases from the upgradient Acme Solvent Site, migration of landfill gas, and a localized source such as effluent from household septic systems in the immediate vicinity.

- A release from Acme Solvent is considered the most likely possibility. The contaminant distributions between the two sites can be explained by the presence of a high permeability zone within the fractured dolomite aquifer between the Acme Solvent Site and the WRL Site. This high permeability zone would result in converging flow "funneling" contaminants into the fracture zone at the upgradient end and diverging flow dispersing contaminants into three dimensions at the downgradient end. Additional studies should be conducted to evaluate this possibility.

- The WRL leachate does not appear to have impacted the surface water or sediments of Killbuck Creek.

- The WRL Site does not appear to have had a significant impact on the ambient air quality.

- A Baseline Risk Assessment was conducted to estimate the risks posed by exposure to chemicals detected in media at the WRL Site. Both noncancer and cancer risks were estimated based on current and potential future Site conditions. Major findings were:

  - Although humans (including sensitive groups such as children) may be potentially exposed to sediments and surface water in Killbuck Creek downgradient of the WRL Site, potential noncancer health effects are not expected and the level of cancer risk is acceptable.

  - Potential future groundwater use represents the greatest risk to humans at the WRL Site. If used as drinking water, the groundwater at the WRL Site may result in an unacceptable level of non-cancer effect and cancer risk. However, contaminants in groundwater upgradient of the WRL Site may contribute in part to this unacceptable health risk.

- An Environmental Assessment was conducted to assess the ecological implication of the levels of chemicals detected in media at the WRL Site. Killbuck Creek and nearby wetlands are sensitive habitats that could potentially be affected by the WRL. Based on the present levels of chemicals detected in the aquatic ecosystem, ecological effects are not expected. Based on the nature of the Site contamination (i.e., groundwater contamination), terrestrial ecosystem effects are not expected.

SECTION 1
INTRODUCTION

1.1 Purpose

This document provides a summary and evaluation of data collected during the Remedial Investigation (RI) performed at the Winnebago Reclamation Landfill (WRL Site or Site), also known as Pagel's Pit Landfill, located near Rockford, Illinois. The primary objectives of the RI are to:

- Determine the extent to which observed releases of volatile organic compounds in the groundwater are attributable, if at all, to the WRL Site.
- Determine if the WRL Site has released contaminants to the groundwater.
- Provide additional definition of the groundwater flow system and groundwater quality between the WRL Site and Acme Solvent Reclaiming, Inc. and determine the source(s) of hazardous releases.
- Perform a risk assessment to evaluate the effects of any known or potential releases from the WRL Site on the public health, welfare, and/or the environment.
- Develop data needed for remedial alternatives evaluation in the Feasibility Study.

1.2 Authorization

This investigation was performed on behalf of the WRL Potentially Responsible Parties (PRPs) who had entered into an Administrative Consent Order with the U.S. EPA, effective October 16, 1986.

The RI was performed following the "Work Plan, Pagel's Pit Landfill Remedial Investigation/Feasibility Study" dated August 1987 and the U.S. EPA "Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA", October 1988. The PRPs retained Warzyn Engineering Inc. (Warzyn) of Madison, Wisconsin (Corporate offices) to perform the tasks in the Remedial Investigation/Feasibility Study (RI/FS) required under the consent agreement.

1.3 Investigative Approach

The RI performed at the WRL Site was accomplished in two phases. Phase I activities were structured to define the nature and extent of potential releases of leachate to the groundwater around the WRL. If required, Phase II work would be conducted, with U.S.EPA approval, to define the migration of groundwater contamination west of Killbuck Creek.

The Phase I activities consisted of the following elements:

- Collection of four rounds of leachate samples from various portions of the facility. Leachate samples were analyzed to evaluate potential hazardous constituents contained in the landfill leachate.
- Installation of 15 new groundwater monitoring wells located adjacent to and downgradient from the WRL Site
- Collection of Rounds 1 and 2 groundwater samples
- Performance of 15 single-well field permeability tests on groundwater monitoring wells, and collection of groundwater levels on an approximately monthly basis
- Collection of Round 1 surface water and Round 1 sediment samples
- Collection of one round of ambient air samples

The purposes of the installation of each Phase I well were as follows:

P3R - Replacement for abandoned water table well P3 to test for releases west of the landfill and to provide data on the relationship between groundwater flow and Killbuck Creek.

P4R - Replacement for abandoned piezometer well P4 having data objectives similar to those for Well P3R.

B15P - Screened 30 feet below the water table to provide water quality data at depth adjacent to Well B15.

G112 - A water table well between the landfill and two impacted water supply wells to the east (G and H, see Drawing 13160-F1). The well may provide an indication if the landfill is impacting the water quality at the water supply wells.

G113, G113A
- Nested wells adjacent to a water supply well (I, see Drawing 13160-F1) which appeared to be impacted, but is not currently functional. Determination of flow direction and groundwater quality may provide an indication of the source of contamination at the former water supply well.

G114 - A water table well adjacent to the landfill in an area of contaminated groundwater. The well may provide an indication as to whether mounding is occurring under the landfill, and the origin of contaminated groundwater in the area.

G115 - A water table well located along the southwest perimeter of the landfill, to monitor for any releases from that portion of the landfill, to test for groundwater mounding, and to provide data on the relationship between groundwater flow and Killbuck Creek.

G116, G116A
- Nested wells placed west of Killbuck Creek to evaluate potential migration of contaminants beneath the creek, and to help characterize the hydraulic relationship between Killbuck Creek and groundwater flow.

G117 - A water table well located adjacent to the landfill and near impacted monitoring wells (MW-106, P1, P7). This well may provide an indication of landfill releases, and data on the hydraulic relationship between groundwater flow and Killbuck Creek.

G118R, G118A
- A nest of wells in an area which appeared to be downgradient of monitoring wells B15 and B15R. These wells will help evaluate the extent of any releases from the landfill.

G119, G119A
- Nested wells to investigate the hydraulic relationship between groundwater flow toward Killbuck Creek, and to assist in the evaluation as to whether there are releases from the landfill.

The August 1987 Work Plan, which outlines the details of the RI, specifies that an Interim Groundwater Quality Evaluation Report (IGQE) be prepared after Phase I. That report included data through the collection of Rounds 1 and 2 groundwater and Round 1 through 3 leachate samples. Based upon comments received on the IGQE

Report, the scope of the Report was expanded to include Round 4 leachate data, resurvey of project wells, and expanded discussions of available background data. The data contained in the IGQE Report was evaluated to determine the following:

- The likely sources of contaminants in groundwater at the western margins of the WRL Site
- The likely sources of contaminants (if any) in groundwater west of Killbuck Creek, a creek located to the west of the landfill
- The likely sources of contaminants to Killbuck Creek, if any
- The necessity for and scope of installation of Phase II groundwater monitoring wells
- The necessity for and details of Round 3 and 4 groundwater sampling and analyses

The evaluation of Phase I groundwater sample results indicated the existing monitoring well network to be adequate to define the extent of releases of hazardous substances from the WRL Site. Therefore, Phase II activities included, with the concurrence of the U.S. EPA:

- Collection of Rounds 3 and 4 groundwater sampling of existing wells
- Collection of Round 2 surface water sampling
- Collection of Round 5 leachate sampling

Additional field permeability testing (12 wells) and water level monitoring (4 events) were also performed.

1.4 Background Information

1.4.1 Setting

The WRL Site is an active solid waste landfill permitted by the State of Illinois and operated by Winnebago Reclamation Services, Inc. (WRS). The WRL Site is located in south central Winnebago County in north central Illinois, approximately 5 miles south of the City of Rockford, in a predominantly rural unincorporated area (Figure 1-1). The WRL Site is located in the east central portion of Section 36, T43N, R1E and the west central portion of Section 31, T43N, R2E of the Third Principal Meridian. The WRL Site is bounded on the west by Killbuck Creek and on the east by Lindenwood Road. Killbuck Creek, a perennial stream, merges with the Kishwaukee River about 2.5 miles northwest of the WRL Site. The Kishwaukee River merges with the Rock River about 1.5 miles northwest of the confluence of Killbuck Creek and the Kishwaukee River. The Site is located on a topographic high between Killbuck Creek to the west and unnamed intermittent streams to the north and south (Figure 1-1).

Surface topography of the Site consists primarily of an area of high relief resulting from the landfill waste disposal operations. The topography surrounding the landfill area is relatively flat to gently rolling. The ground surface ranges from approximately elevation 790 ft mean sea level (MSL) on top of the landfill to approximately 706 ft MSL at Killbuck Creek just west of the landfill (Figure 1-3). A small leachate collection pond is located on top of the landfill.

The landfill lies outside of the 100-year floodplain of Killbuck Creek, and is not within any designated wetland areas. Figure 1-2 shows the 100-year flood boundary and elevations of the 100-year base flood for Killbuck Creek. The floodplain was established by the Federal Emergency Management Agency, and field verified by the Illinois State Water Survey.

A small wetland area south of the landfill (Figure 1-2) was delineated by Encap Inc., on November 5, 1990. This 3.73-acre jurisdictional wetland is rated low in quality because of the artificial nature of its three district portions:

- Scrub-Shrub/Forested Wetland - This is the largest portion of the wetland (3.18 acres), which actually has been created by former and current land use. It is an isolated wetland that happens to be near the creek.

- Shallow Drainway - This portion of the wetland (0.46 acres) is a shallow intermittent ditch that was cut to facilitate drainage of the altered upland and to prevent runoff from entering the adjacent crop field.

- Farmed Wetland - This portion of the wetland (0.09 acres) exists at the terminal end of a natural drainway.

Land use around the WRL Site is a mix of industrial, agricultural, commercial, and rural residential. Figure 1-3 is a current Site features map for the WRL Site. An active sewage sludge drying plant is located on the Site just north of the landfill, and is operated by NRG Technologies, Inc. (NRG).

East of the WRL Site is the former Acme Solvent Reclaiming Inc. (Acme Solvent) NPL Site. The Acme Solvent Site is situated on approximately 20 acres (Figure 1-1), and was used for the disposal of drummed wastes into unlined lagoons and drum stockpiling. The Acme Solvent Site operated from 1960 to 1973. The type, origin, and quantities of wastes disposed at the Acme Solvent Site are generally undocumented, but are known to have included solvent still-bottom sludges, nonrecoverable solvents, paints, and oils. The Illinois Environmental Protection Agency (IEPA) indicates that four lagoons were actively used for the disposal of wastes at the Acme Solvent Site. The IEPA also reported that 10,000 to 15,000 drums may have been present at the Acme Solvent Site when it closed. The total quantity of wastes disposed of at the Acme Solvent Site during its operation is unknown (Ecology and Environment, 1983; Jordan, 1984). IEPA inspections in late 1972 and early 1973 indicated the wastes in solvent lagoons at the Acme Solvent Site were not removed, but were covered with soil. It was also reported that an unknown number of on-site drums were crushed and buried, rather than removed (Ecology and Environment, 1983). A partial clean-up and removal of buried drums and contaminated soils from the Acme Solvent Site began in August 1986.

The Rockford Skeet Club is across Lindenwood Road to the northeast of the WRL Site. A septic tank pumping business is located to the west, a private hunt club to the southwest, and a limestone quarry to the east of the WRL Site. There are scattered residences within 1/2 mile of the Site to the north, south, southwest, and southeast. Of these, only one residence, located near private well PW1 (also identified as PWO) (see Drawing 13160-F1), is hydraulically downgradient of the WRL Site.

1.4.2 Site History and Operations

The landfill has been in operation since 1972 and although the landfill has an estimated 5 to 7 years of capacity remaining at current filling rates, WRS has notified the state that operations will continue beyond seven years. The landfill has an asphalt liner and a leachate collection system. A system of gas wells are used to extract landfill gas, which is used as a fuel for the municipal sludge dryer. Wastes accepted at the WRL Site are composed primarily of municipal refuse and sewage treatment plant sludge from the Rock River Water Reclamation District (RRWRD) City of Rockford sewage treatment plant. The landfill accepted sewage treatment plant sludge until January 1985. Since January 1985, only dried sludge has been placed in the landfill. A very limited amount of Illinois special non-municipal wastes were disposed at the facility prior to December 1975 under permits issued by the IEPA. Not all of the special wastes permitted by the IEPA were actually disposed of at the landfill (WRS, 1984).

1.5 Previous Investigations

The area at the WRL Site and the Acme Solvent Site facility has been investigated a number of times. The previous investigations are:

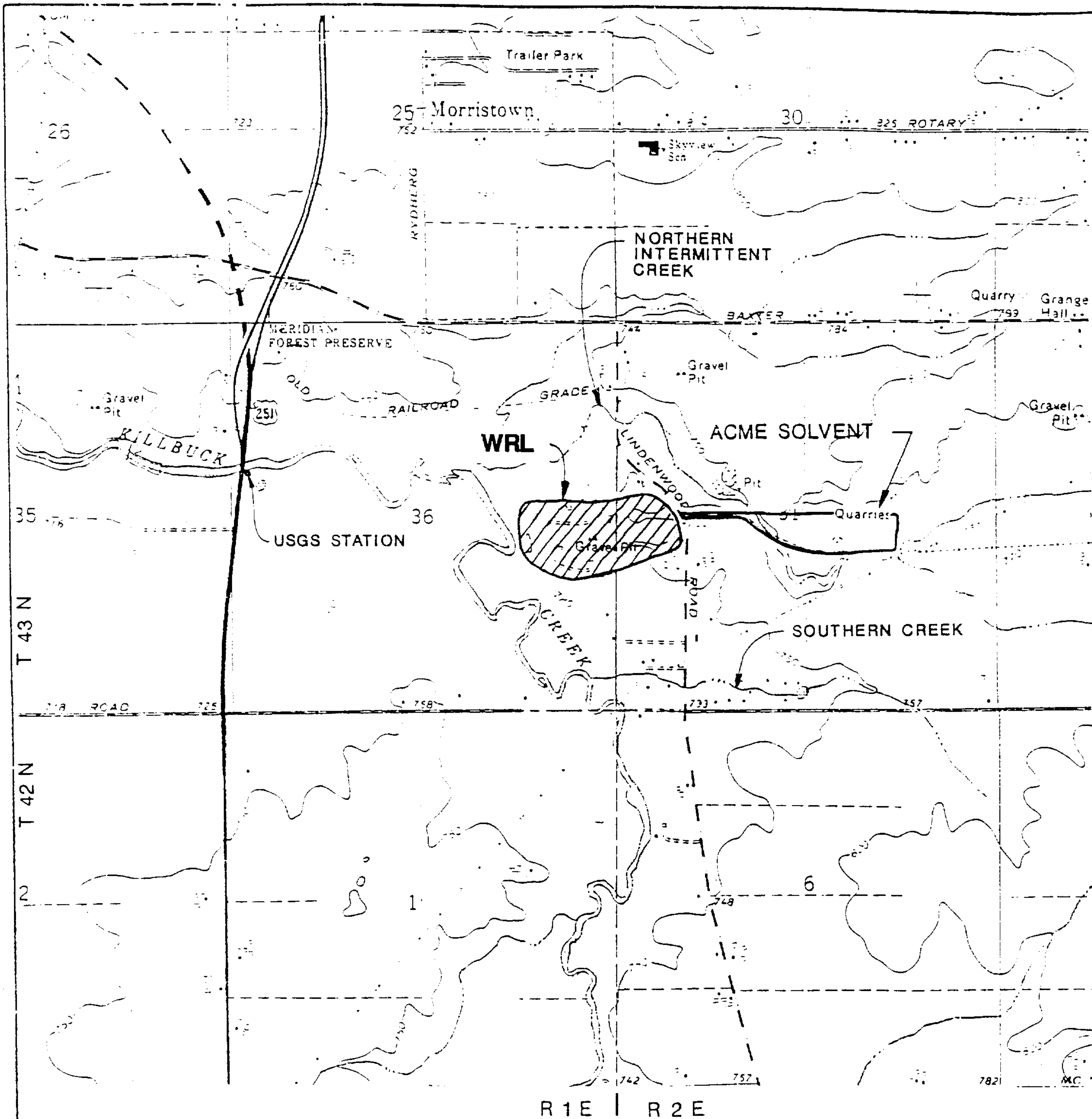
- "Extent of Source of Groundwater Contamination - Acme Solvents Pagel Pit Area near Morrisville, Illinois", Ecology and Environment, March 1983.
- "Acme Solvents Superfund Site, Winnebago County, Illinois, Remedial Investigation", E.C. Jordan Company, September 1984.
- "Acme Solvents Superfund Site, Winnebago County, Illinois, Feasibility Study", E.C. Jordan Company, February 1985.
- "Supplemental Investigation Winnebago Reclamation Landfill, Rockford, Illinois", Warzyn Engineering Inc., March 1985.
- "Data Analysis and Summary Report for Deep Groundwater Assessment, Acme Solvents Superfund Site", E.C. Jordan Company, May 1986.

Monitoring wells installed for these previous investigations were incorporated into the Phase I RI at the WRL Site. At the completion of the Phase I RI, the IGQE Report was prepared by Warzyn. The data collected during previous report efforts were considered in the evaluation of Site conditions in that document. Complete data sets from previous reports were not incorporated into the document, but were referenced where appropriate.

During previous investigations, groundwater samples were collected on or about October 25, 1982, September 6, 1983, May 1, 1984, December 27-29, 1984, January 2-4, 1985, and November 5, 1985. The October 1982 samples were collected by Ecology and Environment from monitoring wells B1 through B16 plus five water supply wells (F, G, H, I, and J on Drawing 13160-F1). The samples were analyzed for priority pollutants (volatile and semi-volatile organics and metals). Ecology and Environment collected groundwater samples in September 1983 from monitoring wells B1 through B16 and six water supply wells (those sampled in 1982 plus well E). The samples were analyzed for volatile and semi-volatile organic compounds (December 28, 1983 memorandum Tom Koch to File). E.C. Jordan Co. collected groundwater samples in May 1984 from G101, G102, MW101 through MW107, B1 through B12, P1, P3, P4, P5, P7, and water supply wells A through O. The samples were analyzed for volatile and semi-volatile organic priority pollutants and PCBs.

In December 1984 and January 1985, Warzyn collected groundwater samples from monitoring wells B8, B9, B10, B10A, B11, B11A, B12, B13, B14, B16, B16A, P1, P3, P4, P5, P6, P7, G102, G107, G108, G109, G109A, G110, G111, MW104, MW106, MW107, and water supply wells G and H. These samples were analyzed for VOCs, arsenic, barium, cadmium, phenols, chloride, and alkalinity. E.C. Jordan (1986) collected samples from wells MW201A, MW201B, and MW202, which were analyzed for volatile and semi-volatile organic priority pollutants.

In late 1989, soil borings SB1-SB5 were performed as part of an evaluation of the shallow stratigraphy for potential landfill expansion south of the WRL Site. The boring logs are included in Appendix F.



BASE MAP DEVELOPED FROM ROCKFORD SOUTH, ILLINOIS
7.5 MINUTE USGS TOPOGRAPHIC QUADRANGLE MAP
DATED 1971 PHOTOREVISED 1976

north

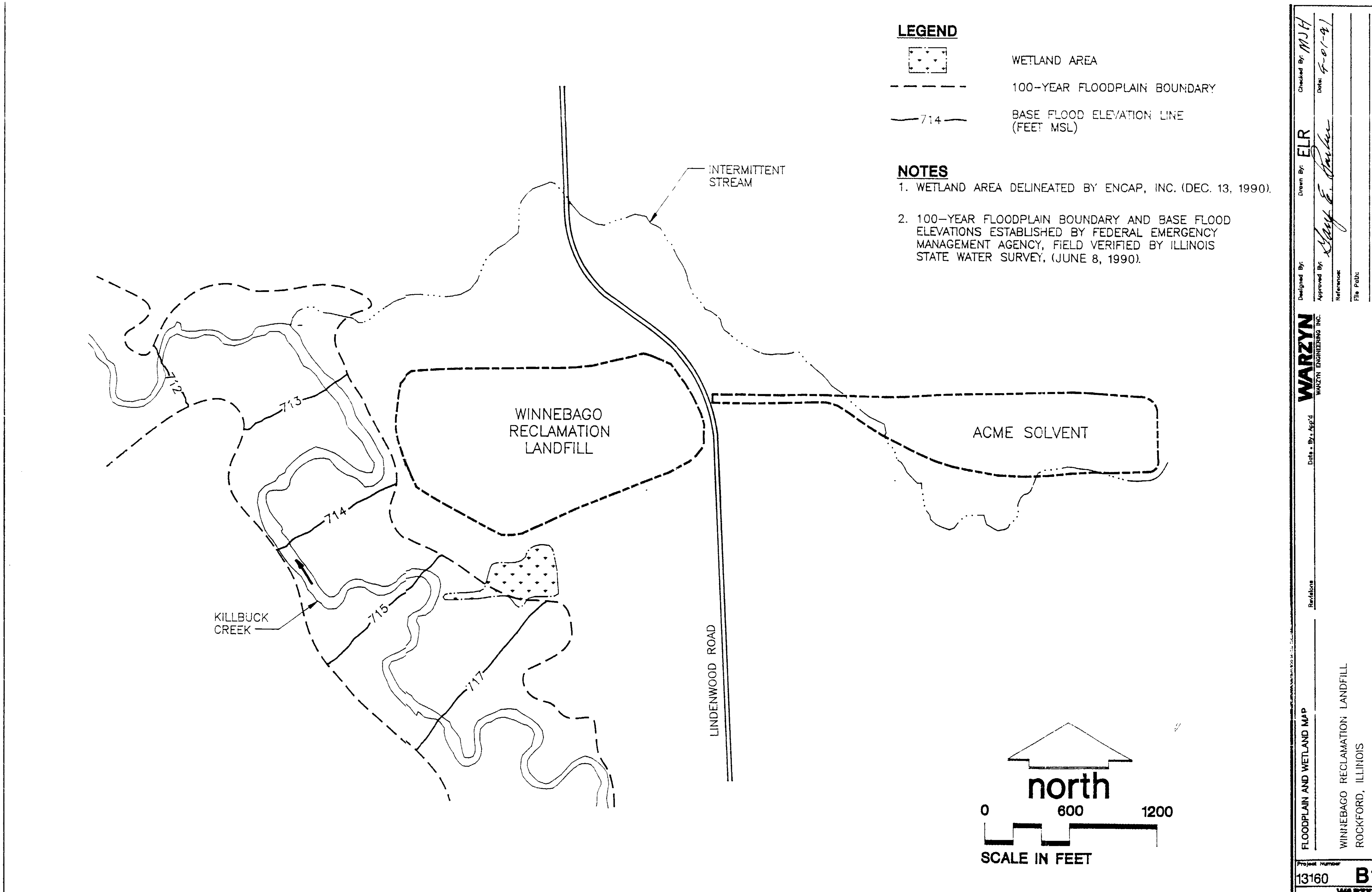
SCALE: 1"=2000'

FIGURE 1-1

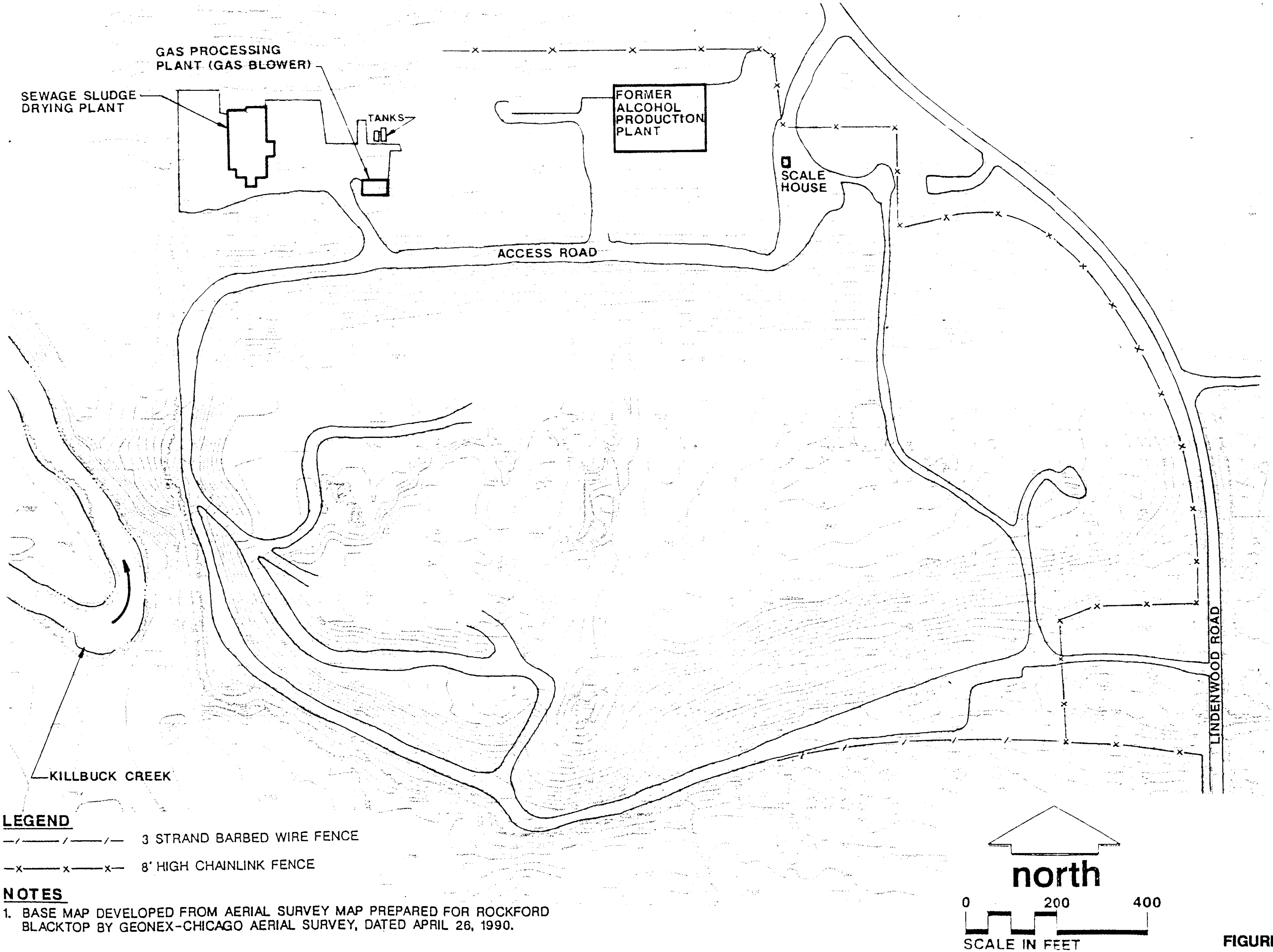
WARZYN ENGINEERING INC	SITE LOCATION MAP	
	WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS	

DWN D.L.L.	APP'D	DATE 11/22/89	13160	A1

TELEDYNE POST N67178



LEGEND
WETLAND AREA
100-YEAR FLOODPLAIN BOUNDARY
714
BASE FLOOD ELEVATION LINE
(FEET MSL)
NOTES
1. WETLAND AREA DELINEATED BY ENCAP, INC. (DEC. 13, 1990).
2. 100-YEAR FLOODPLAIN BOUNDARY AND BASE FLOOD ELEVATIONS ESTABLISHED BY FEDERAL EMERGENCY MANAGEMENT AGENCY, FIELD VERIFIED BY ILLINOIS STATE WATER SURVEY, (JUNE 8, 1990).
INTERMITTENT STREAM
WINNEBAGO RECLAMATION LANDFILL
ACME SOLVENT
LINDENWOOD ROAD
KILLBUCK CREEK
712
713
714
715
717
north
0
600
1200
SCALE IN FEET
Drawn By: ELR
Checked By: MJH
WARZYN
WARZYN ENGINEERING INC.
FLOODPLAIN AND WETLAND MAP
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
Project Number
13160
B1
WARZYN



LEGEND

—/——/——/— 3 STRAND BARBED WIRE FENCE

—x——x——x— 8' HIGH CHAINLINK FENCE

NOTES

1. BASE MAP DEVELOPED FROM AERIAL SURVEY MAP PREPARED FOR ROCKFORD BLACKTOP BY GEONEX-CHICAGO AERIAL SURVEY, DATED APRIL 26, 1990.

FIGURE 1-3

CURRENT SITE FEATURES MAP

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

WARZYN

Drawn by: DLL
Checked: MJH
Approved by: Gary E. Parker
Date: 4-01-91

13160 B2

WARZYN

SECTION 2
DATA COLLECTION ACTIVITIES

This section provides a brief summary of work performed during the Remedial Investigation. The activities were performed in a manner consistent with the approved Work Plan, Health and Safety Plan, Sampling Plan, and Quality Assurance Project Plan developed for the investigation.

2.1 Phase I Data Collection Activities

2.1.1 Phase I Monitoring Well Installation

During Phase I, 15 new groundwater monitoring wells were installed adjacent to the WRL Site. The locations of Phase I groundwater monitoring wells; P3R, P4R, B15P, G112, G113, G113A, G114, G115, G116, G116A, G117, G118R, G118A, G119, and G119A are shown on Drawing 13160F1. Monitoring wells P3R and P4R were installed to replace wells P3, P4, and P5, abandoned as landfill construction advanced to the west. Monitoring well G118 had to be replaced, due to frost heave of the casing at the surface. The replacement well, installed on June 9, 1988, was designated G118R.

Drilling and monitoring well installation were performed by Exploration Technology, Inc. (ETI) of Madison, Wisconsin and observed by Warzyn personnel. A Warzyn geologist was on-Site during drilling operations to log boreholes, provide technical supervision, and perform air monitoring related to health and safety concerns.

The deepest borehole at a nested well location was sampled at 5-ft intervals in unconsolidated material to bedrock or the terminus of the boring. Unconsolidated materials were collected using a split-spoon sampler (ASTM Method D-1586). When bedrock was encountered, rock was cored to the terminus of the boring using a HW (2-7/8-inches diameter) core barrel.

The monitoring wells installed were constructed with 2-in. inside diameter stainless steel screen and riser pipe to an elevation above the anticipated seasonal high groundwater elevation. Galvanized steel riser pipe was used above the assumed high groundwater elevation to the ground surface. Wells installed to monitor the groundwater surface (water table wells) were constructed with 10-ft screens spanning the anticipated water level fluctuations. Monitoring wells P3R, G112, G113, G114, G115, G116, G117, G118R, and G119 were instrumented as water table wells. Wells instrumented to monitor groundwater quality and groundwater levels deeper in the aquifer (piezometers) were installed with 5-ft well screens. Monitoring wells P4R, B15P, G113A, G116A, G118A, and G119A were instrumented as piezometers.

The annular space between the well materials and the edge of the borehole was backfilled to approximately 2-ft above the top of the screen with flint sand. Above the flint sand, a 2-ft bentonite pellet seal was placed. Bentonite grout was placed via a tremie pipe to fill the remaining annular space to the surface. At the ground surface, locking protective casings were installed.

To minimize possible contamination of monitoring wells, the drill rig and drilling tools were decontaminated by steam-cleaning at the start of drilling and prior to use at each drilling location. The split-spoon sampler and/or core barrel were cleaned between samples using a trisodium phosphate wash followed by clean water rinses. Well construction materials were also steam-cleaned prior to use.

Appendix A contains boring logs for borings performed as part of the Phase I investigation. Appendix B contains well installation details for monitoring wells installed as part of Phase I.

2.1.2 Groundwater Sampling

The initial set of Phase I groundwater samples was collected the week of April 6, 1988 (Round 1), with a second set of samples collected the week of June 9, 1988 (Round 2). Samples collected from the two Acme Solvent Site monitoring wells (STI-5I and STI-5D) could not be obtained concurrent with the other samples, since the wells were not installed when sampling occurred. See Section 2.3 for details.

Analytical laboratory results for groundwater analyses are contained in Appendix C. Monitoring well locations are shown on Drawing 13160-F1.

2.1.2.1 Round 1 Groundwater Sampling. Monitoring wells B15, B15R, B15P, G109, G109A, G115, G116, G116A, G117, G118A, MW106, P1, P3R, and P4R were sampled and analyzed in Round 1 for the following parameters:

- Volatile organic compounds (VOCs) by gas chromatograph (GC). Ten percent of the samples were confirmed by gas chromatography/mass spectroscopy (GC/MS)
- Semi-volatile organic compounds (SVOCs)
- Pesticides/PCBs
- 23 metals (dissolved)
- Total phenolics
- Field pH
- Field specific conductance
- Total alkalinity
- Chlorides
- Cyanide

G118 was not sampled, because it was damaged by frost heave.

A second set of wells which included: B4, B6S, B6D, B7, B9, B10, B11, B11A, B12, B13, B14, B16, B16A, G107, G108, G110, G112, G113, G113A, G114, G119, G119A, P6, P7, and MW105, and a private well (PW1), were sampled for indicator parameters, as follows:

- VOCs (by GC) with 10% GC/MS confirmation
- Total phenolics
- Field pH
- Field specific conductance
- Total alkalinity
- Chlorides
- Arsenic (dissolved, except PW1, unfiltered)

- Barium (dissolved, except PW1, unfiltered)
- Cadmium (dissolved, except PW1, unfiltered)

2.1.2.2 Round 2 Groundwater Sampling. Wells B4, B6S, B6D, B7, B9, B10, B11, B11A, B12, B13, B14, B15, B15R, B15P, B16, B16A, G107, G108, G109, G109A, G110, G111, G112, G113, G113A, G114, G115, G116, G116A, G117, G118A, G118R, G119, G119A, P1, P3R, P4R, P6, P7, MW105, MW106, and a private well (PW1/PWO), were sampled during Round 2. Samples from these wells were analyzed for the indicator parameters described in Section 2.1.2.1. See Appendix C for results of Round 2 sampling.

2.1.3 Leachate Sampling

Four sets of leachate samples were collected during Phase I. Leachate sample Rounds 1 and 2 were collected concurrent with Round 1 and Round 2 groundwater sampling. Round 3 leachate samples were collected during the week of August 8, 1988, and Round 4 on June 8, 1989. On each occasion, an attempt was made to collect six leachate samples. Two of the six samples in each set were from leachate extraction manholes, which are tied to the base leachate collection piping system. The remaining four samples in a set of six samples were collected from various gas extraction wells located in the eastern portion of the Site. Figure 2-1 shows the locations of the manholes and gas extraction wells. Leachate samples were collected from the following locations:

- Round 1 - West Manhole, A1, E7, F1, K2, (East Manhole dry)
- Round 2 - East and West Manholes, A3, B1, G3, G7
- Round 3 - C3, C6, H2, I5, (both manholes dry)
- Round 4 - D1, G4, F7, East Manhole, (West Manhole dry, C6 dry)

Leachate samples were not filtered prior to analysis. Round 1 leachate samples were analyzed for VOCs by GC (with one sample confirmed by GC/MS), SVOCs, 23 metals, chlorides, alkalinity, total phenolics, cyanide, field pH, field specific conductance, and Pesticides/PCBs. Rounds 2 and 3 samples were analyzed for VOCs by GC, arsenic, barium, cadmium, chlorides, alkalinity, total phenolics, field pH, and field specific conductance. One Round 1 sample and one Round 2 sample had VOCs confirmed by GC/MS. Due to analytical difficulties (matrix interference) noticed in Rounds 1 through 3 results, Round 4 leachate samples were analyzed by GC/MS for VOCs. Round 4 samples were also analyzed for alkalinity, chloride, total phenolics, field pH, field specific conductance, arsenic, barium, and cadmium. Analytical results are contained in Appendix C.

Round 1, 2, and 3 leachate samples were collected from the specified gas extraction wells by taking the well off-line, disconnecting it from the extraction system, and collecting the sample with a stainless steel bailer (no leachate volumes were purged prior to sample collection). During leachate sampling Rounds 4 and 5, the sample collection procedure was modified to include taking the gas extraction wells to be sampled off-line and pumping the wells dry 24-hours prior to sampling. Sample collection then proceeded as for Rounds 1, 2, and 3. Samples were collected directly from the leachate manholes, without purging during each sampling round.

2.1.4 Surface Water and Sediment Sampling

Surface water (SW) and sediment (SD) samples were collected from five locations along Killbuck Creek in the immediate vicinity of WRL (see Drawing 13160-F1). Sampling proceeded from the furthest downstream location (SW1/SD1) toward the furthest upstream location (SW5/SD5). At each sample location, the surface water sample was collected first, followed by the collection of the sediment sample.

Sediment samples were analyzed for VOCs (by GC/MS), SVOCs, Pesticides/PCBs, 23 metals, and cyanide. Surface water samples were analyzed for VOCs (by GC with one sample confirmed by GC/MS), arsenic, barium, cadmium, field specific conductance, field pH, chloride, alkalinity, and total phenolics. Surface water samples were not filtered prior to analysis. Results of sediment and surface water sampling and more detailed parameter lists are contained in Appendix C.

2.1.5 Permeability Testing

Permeability testing was performed on monitoring wells; B11, B11A, B15, B15P, B16, B16A, G109, G109A, G111, G115, G116, G116A, G117, G119, and G119A. Two different permeability test procedures were used for piezometer wells and water table wells, respectively. Piezometers were tested by suppressing the water level using a compressed air system. Rapid release of pressure in the wells allowed the suppressed water levels to recover to their original levels. Recovery of groundwater levels were recorded using a pressure transducer system attached to a digital data logger. Water table wells had a small volume of water removed using a bailer (baildown method); recovery was recorded using a pressure transducer and digital data logger. Results of field permeability testing are summarized in Table 2-1, with detailed data assessments provided in Appendix D. Results were evaluated using the method of Bouwer and Rice (1976).

2.1.6 Survey

Project wells were resurveyed in August 1989 by an Illinois Registered Land Surveyor. The horizontal location (Illinois State Plane Coordinates, West Zone) of the wells were surveyed to an accuracy of $\pm$0.1 ft. Elevations referenced to mean sea level (MSL) (National Geodetic Vertical Datum) were obtained for the top of the protective casing (TOC) and inner casing (well pipe) (TIC) to an accuracy of $\pm$0.01 ft. The ground surface elevations were surveyed to an accuracy of $\pm$0.1 ft. Elevation data are provided in Appendix B with well locations shown on Drawing 13160-F1.

2.1.7 Water Level Monitoring

The variable groundwater flow patterns noted in previous investigations near WRL necessitated the collection of multiple sets of groundwater levels from functioning groundwater wells to further understand the flow dynamics. Groundwater levels were obtained on the following dates:

- April 6, 1988
- May 17, 1988
- June 16, 1988
- August 5, 1988

- August 29, 1988
- October 25, 1988
- November 9, 1988
- June 8, 1989

Water level data are included in Appendix E.

Water level measurements have precision and accuracy much like any other measurement. Random inaccuracies occur when the measurements are recorded or taken incorrectly, potentially resulting in anomalous data. Systematic inaccuracies may result from the survey data or the measuring device. The reference point (top of inner well casing) is surveyed to +/- 0.01 ft, and the water level measurements are made to +/- 0.01 ft, for an accuracy of +/- 0.02 ft. When comparing the measurements from nested wells to determine vertical gradients, the accuracy is +/- 0.04 ft. The combined accuracy of the measurements must be considered when comparing water level measurements of nested wells to determine if their differences are greater than the accuracy of the measurements themselves (i.e., is the difference in the water level elevations meaningful).

2.1.8 Ambient Air Monitoring

A single event, ambient air monitoring was conducted on October 24-25, 1988 at the WRL Site. Samples were collected for analysis of VOCs on Terrax/Terrax Charcoal sorbent tubes supplied by Radian Corporation.

Alpha II personal air sampling pumps were placed at three downwind sampling locations (Downwind #1, #2, and #3) and one upwind (Upwind #4) sampling location (Figure 2-2), and a meteorological station was established to monitor wind speed, wind direction, and air temperature (Appendix C). Sampling pumps were calibrated to approximately 20 cc/minute flow rates. Three (3) duplicates were taken at Downwind #2 at flow rates of approximately 10, 30, and 40 cc/minute. The sampling pumps were calibrated and programmed for a run time of 12 hours (Appendix C).

Sorbent tube #2507 used at downwind location #1 was broken during the sampling event when the tripod tipped over and was not analyzed. Sorbent tube #2508 was broken during transport prior to the air sampling and was not analyzed. Analytical laboratory results are contained in Appendix C.

2.2 Phase II Data Collection Activities

2.2.1 Groundwater Sampling

Two rounds of groundwater sampling were performed during Phase II of the RI. Round 3 groundwater sampling was initiated on February 5, 1990. Round 4 groundwater sampling commenced on April 23, 1990. Analytical results of these sampling events are presented in Appendix C. Locations of sampled monitoring wells are illustrated in Drawing 13160-F1.

2.2.1.1 Round 3 Groundwater Sampling. Monitoring wells B13, B15R, B15P, G109, G109A, G110, G111, G114, G115, G116, G116A, P1, P3R, P4R, and P6 were sampled and analyzed during Round 3 groundwater sampling. The parameters analyzed included:

- VOCs by GC/MS
- SVOCs
- Total phenolics
- Dissolved arsenic
- Dissolved barium
- Dissolved cadmium
- Total chromium (dissolved)
- Hexavalent chromium
- Total cyanide
- Total alkalinity
- Chloride
- Sulfate
- Nitrate plus nitrite nitrogen

- Field pH
- Field specific conductance

2.2.1.2 Round 4 Groundwater Sampling. During Round 4 groundwater sampling, monitoring wells B13, B15R, B15P, G109, G109A, G110, G111, G114, G115, G116, G116A, P1, P3R, P4R, and P6 were sampled and analyzed for the following parameters:

- VOCs by GC/MS
- Total phenolics
- Dissolved arsenic
- Dissolved barium
- Dissolved cadmium
- Total cyanide
- Total alkalinity
- Sulfate
- Chloride
- Field pH
- Field specific conductance

2.2.2 Leachate Sampling

Round 5 leachate samples were collected during Phase II of the RI. Sampling procedures used for Round 4 were followed. Sampling locations included the east manhole, west manhole, and gas extraction wells D1, E3, N8, and L2. Sampling locations are illustrated in Figure 2-1. Leachate samples were not filtered prior to analysis. Analytical results are summarized in Appendix C.

Round 5 leachate samples were analyzed for the following parameters:

- Alkalinity

- Chloride
- Total phenolics
- Sulfate
- Nitrate plus nitrite nitrogen
- 23 metals
- Cyanide
- Hexavalent chromium
- Pesticides and PCBs
- VOCs by GC/MS
- SVOCs
- Field pH
- Field specific conductance

2.2.3 Surface Water Sampling

Round 2 surface water samples were collected at the five locations previously sampled during Phase I of the RI. These sampling locations are illustrated on Drawing 13160-F1. Round 2 surface water samples were analyzed for VOCs (GC/MS), total alkalinity, chloride, total phenolics, arsenic, barium, cadmium, total chromium, hexavalent chromium, total cyanide, field pH, and field-specific conductance. Surface water samples were not filtered prior to analysis. Analytical results of the Round 2 surface water sampling are presented in Appendix C.

2.2.4 Permeability Testing

Permeability testing was performed on 12 additional monitoring wells during Phase II, utilizing the baildown method described in section 2.1.5. The wells tested included B9, B10, B10A, B12, B13, G108, G109A, G110, G113, G113A, G114, and P6. Results of the baildown analyses are summarized in Table 2-1, with detailed data analyses provided in Appendix D. Results were evaluated utilizing the method of Bouwer and Rice (1976).

2.2.5 Water Level Monitoring

During Phase II of the RI, groundwater levels were obtained on the following dates:

- December 8, 1989
- February 5, 1990
- March 5, 1990
- April 23, 1990

Water levels are summarized in Appendix E.

2.3 Variations from the QAPP

Rounds 3 and 4 groundwater samples were analyzed utilizing the GC/MS methodology, instead of the GC methodology with 10% GC/MS confirmation. SVOCs, total cyanide, sulfate, nitrate plus nitrite nitrogen, total chromium (dissolved), and hexavalent chromium were added to the Round 3 parameter list. Total cyanide and sulfate were added to the Round 4 groundwater parameter list.

Round 4 leachate sampling occurred under Phase I, instead of Phase II. Rounds 4 and 5 leachate samples were analyzed for VOCs using GC/MS, instead of GC with 10% confirmation using GC/MS. The sampling procedure was changed to purging leachate from the gas well 24 hours prior to collection of the leachate sample. During the 24-hour period, the gas well was open to the atmosphere to prevent the build-up of dangerous levels of landfill gas. Sulfate, nitrate plus nitrite nitrogen, SVOCs, pesticides/PCBs, 23 metals, cyanide, and hexavalent chromium were added to the Round 5 parameter list. The gas well leachate sampling locations were changed to address the problem of dry wells (and retesting of D1 in Round 5).

Round 2 surface water samples were tested for the following additional parameters: total chromium, hexavalent chromium, and total cyanide. In addition, the VOCs were analyzed using GC/MS instead of GC with 10% GC/MS confirmation.

The Hydrologic Evaluation of Landfill Performance (HELP) Model (Version 2.05) was used in the Feasibility Study to evaluate the effectiveness of the cap/cover alternatives. The HELP model was not used to compare leachate collected to potential leachate volumes due to data limitations.

Samples from wells STI-5I and STI-5D were to be collected concurrent with Rounds 1 and 2 groundwater sampling, however, the wells were not installed at that time. On August 10, 1988, Warzyn obtained a split-sample from the Harding Lawson Associates (HLA) sampling crew from well STI-5I. The STI-5I sample was analyzed for 23 metals, cyanide, and total phenolics. HLA was unable to sample well STI-5D due to a pump malfunction, so Warzyn could not obtain a split-sample. On November 12, 1988, Warzyn obtained a split-sample from the HLA sampling crew from well STI-5D which was analyzed for cyanide and total phenolics. HLA was unable to sample STI-5I that same day, so a set of sample bottles and shipping materials was left with the oversight contractor to obtain the split-sample when possible. The sample from STI-5I was obtained on November 14, 1988 and analyzed for cyanide and total phenolics.

In addition to the permeability testing performed in Phase I, field single-well permeability tests were performed on 12 more monitoring wells.

Table 2-1

Hydraulic Conductivity Test Results (cm/s)

Dolomite Bedrock

1988 Warzyn Results		1990 Warzyn Results		1984 E.C. Jordan Results	
B11	$1x10^{-3}$	B9	$4.5x10^{-5}$	MW105	$1x10^{-4}$
B11A	$1.2x10^{-4}$	B10	$7.2x10^{-5}$	MW107	$4.4x10^{-5}$
B15P	$5.4x10^{-4}$	B10A	$1.9x10^{-3}$	P8	$3x10^{-4}$
B16	$8.2x10^{-4}$	B12	$5.6x10^{-5}$	P9	$1.5x10^{-4}$
B16A	$1.6x10^{-4}$	B13	$1.4x10^{-3}$		
G109	$3.2x10^{-4}$	G108	$2.8x10^{-3}$		
G109A	$4x10^{-6}$	G109A	$5.4x10^{-6}$		
G111	$2.4x10^{-2}$	G110	$1.4x10^{-3}$		
		G113	$1.4x10^{-4}$		
		G113A	$5.4x10^{-5}$		
		G114	$6.6x10^{-5}$		
		P6	$2x10^{-3}$		

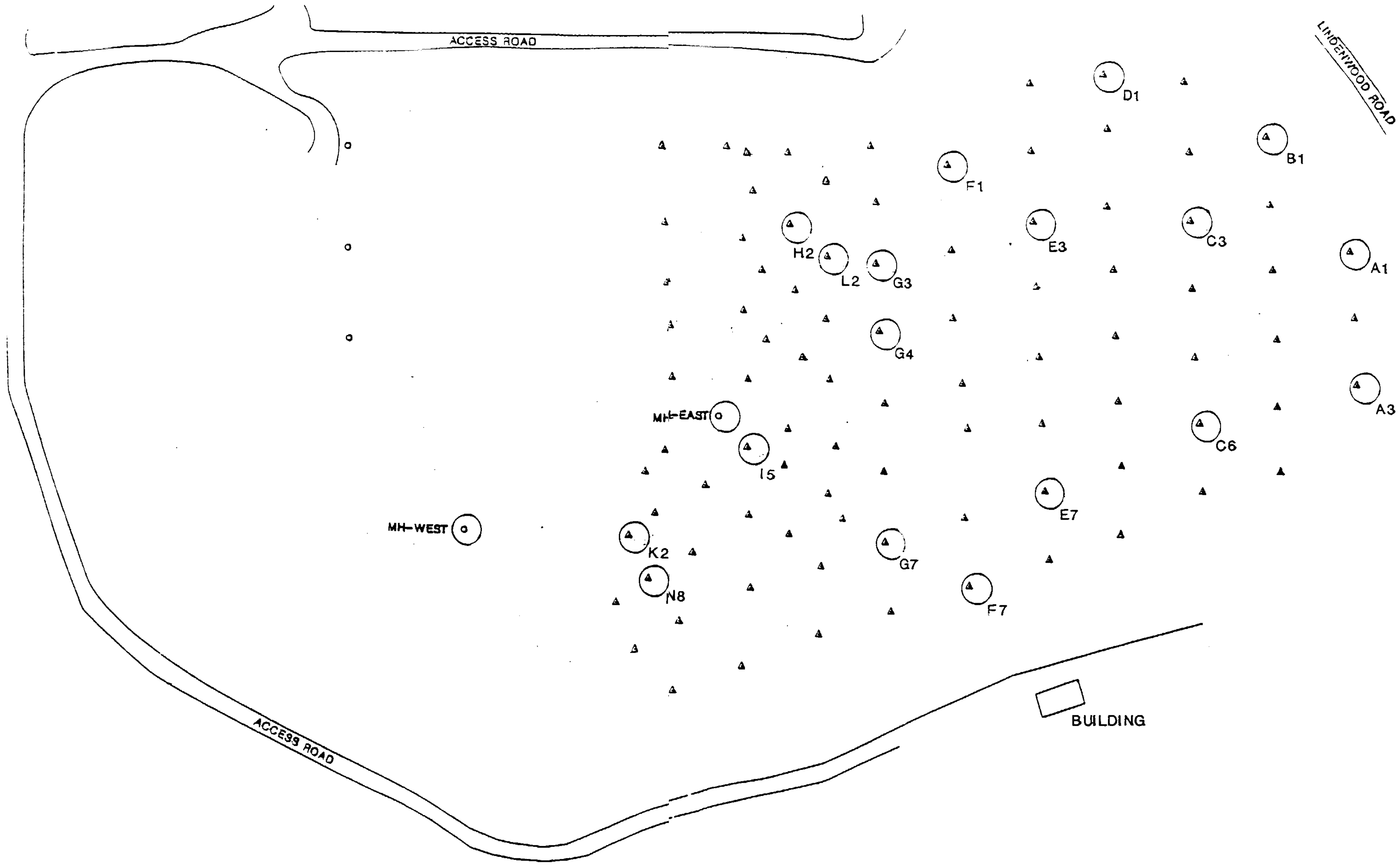
Bedrock Geometric Mean[1] = $2.3x10^{-4}$ cm/sec

Unconsolidated Sediments

B15	$3.7x10^{-1}$	G117	$2x10^{-4}$
G115	$1.1x10^{-2}$	G119	$5.7x10^{-2}$
G116	$7.5x10^{-3}$	G119A	$2.8x10^{-3}$
G116A	$8x10^{-4}$		

Unconsolidated Geometric Mean[1] = $7x10^{-3}$ cm/sec

1 Geometric Mean = exp [1/n ((ln(X1) + ln(X2) +...+ ln (Xn))]
(Sudicky 1986; Gilbert 1987)



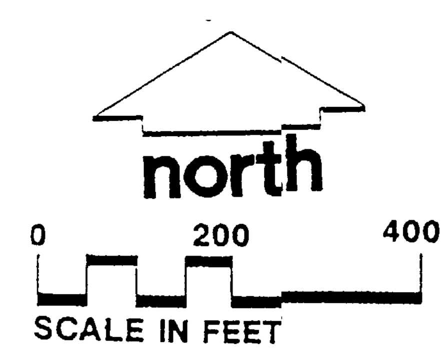
LEGEND

- △ GAS WELL/LEACHATE EXTRACTION POINT
- ○ LEACHATE COLLECTION SYSTEM MANHOLE
- (△) A1 EXTRACTION WELLS SAMPLED

ROUND #	WELLS SAMPLED
ROUND 1	WEST MANHOLE, A1, E7, F1, K2, (EAST MANHOLE DRY)
ROUND 2	BOTH MANHOLES, A3, B1, G3, G7
ROUND 3	C3, C6, H2, I5, (BOTH MANHOLES DRY)
ROUND 4	D1, G4, F7, EAST MANHOLE, (WEST MANHOLE & C6 DRY)
ROUND 5	BOTH MANHOLES, D1, E3, L2, N8

NOTES

1. BASE MAP TAKEN FROM GAS WELL LOCATION MAP DATED OCT. 30, 1989, PROVIDED BY WRL.



WARZYN
WARZYN ENGINEERING INC.

LEACHATE SAMPLING LOCATION MAP

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

Project Number 13160 B5

LEGEND

APPROXIMATE LANDFILL BOUNDARY

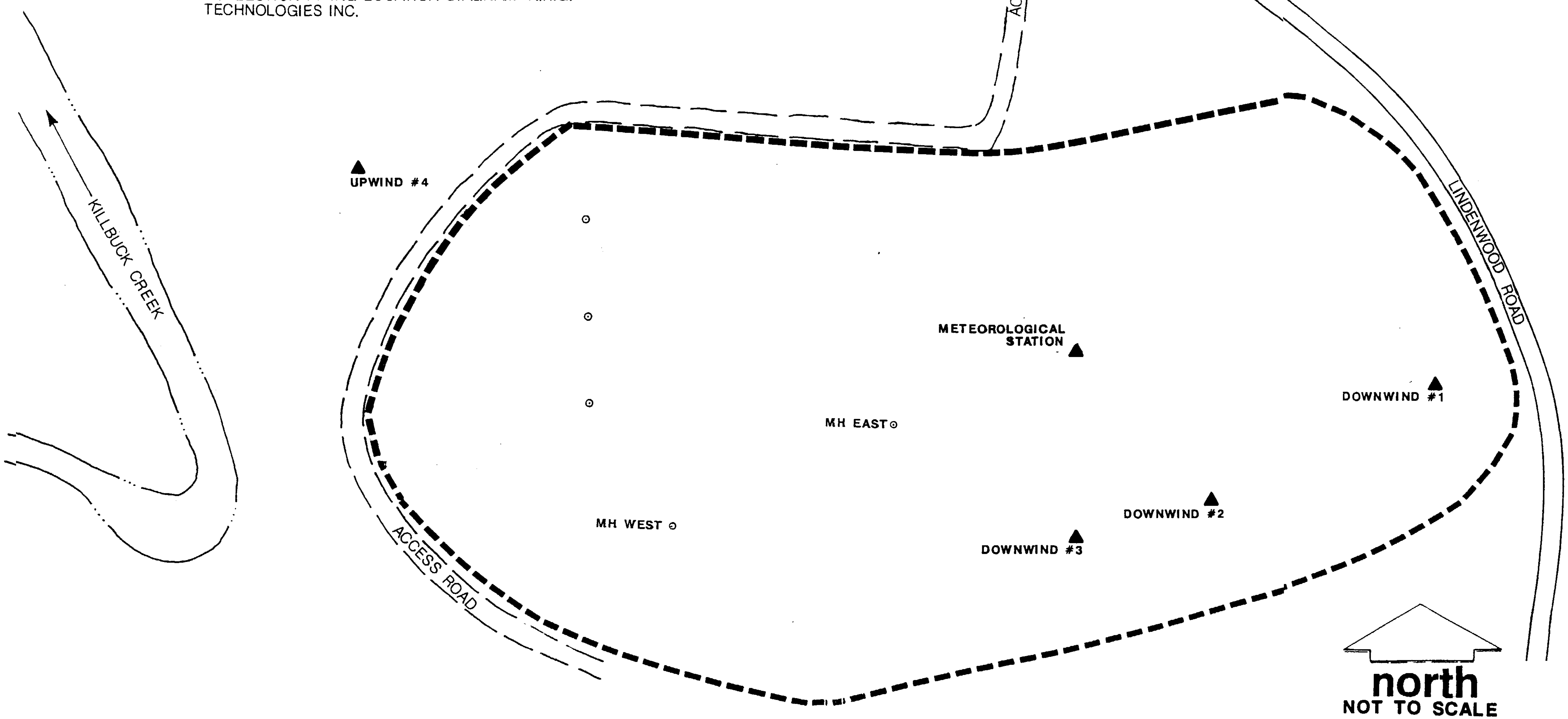
MANHOLE LOCATION

AIR SAMPLING LOCATION

NOTES

1. BASE MAP DEVELOPED FROM DWG. NO. C12660-B3, "RI SAMPLING LOCATIONS," WINNEBAGO RECLAMATION SERVICE LANDFILL.

2. GAS WELL LOCATIONS TAKEN FROM "GAS WELL COLLECTION PIPING LOCATION DIAGRAM" N.R.G. TECHNOLOGIES INC.



AIR SAMPLING LOCATION MAP

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

WARZYN
WARZYN ENGINEERING INC.

Drawn By: D.L.L. Checked By: CWB
Date: 4/01/91

Project Number 13160 B6

SECTION 3
PHYSICAL SETTING

The WRL Site is located in the Rock River Hill Country of the Till Plains Section of the Central Lowland Province of Illinois (Figure 3-1) (Leighton, et al., 1948). The Rock River Hill Country is characterized by subdued rolling hills rising above alluvial valleys. An extensive surface drainage system has developed in the uplands, making lakes, ponds, and marshes rare. Major river valleys are broad with steep walls and alluvial terraces (Hackett, 1960, Leighton, et al., 1948).

The WRL Site is located on a topographic high between Killbuck Creek to the west, and unnamed intermittent streams to the north and south (Figure 1-1). Killbuck Creek, a perennial stream, flows within 250 ft of the western boundary of the WRL Site and merges with the Kishwaukee River about 2.5 miles to the northwest. The confluence of Killbuck Creek and the northern intermittent stream is about 1000 ft northwest of the Site, and the confluence of the southern intermittent unnamed stream and Killbuck Creek is about 1200 ft south of the WRL Site. The ground surface ranges from approximately elevation 790 ft MSL on top the the landfill to approximately 706 ft MSL in the floodplain of Killbuck Creek (Figure 1-3). There are no other significant surface water bodies within 1 mile of the WRL Site.

The average precipitation for the area is 38 in. per year; 66% being received between April and September, with an average snowfall of 33 in. In winter, the average temperature is 23°F, and the average summer temperature is 71°F (USDA, 1980).

3.1 Regional Geology

3.1.1 Unconsolidated Materials

The surficial unconsolidated materials of the area are predominantly glacial drift deposits. Glacial drift deposits include both ice and water-lain materials. Figure 3-2 is a geologic map of the surficial unconsolidated materials in the vicinity of the WRL Site. The area beneath and east of the Site is mapped as poorly-sorted sand and gravel glacial ice-contact deposits of the Wasco Member of the Henry Formation. West of the Site in the

Killbuck Creek Valley, and to the north of the Site, the surficial sediments are mapped as the sand and gravel outwash deposits of the Mackinaw Member of the Henry Formation. Modern stream deposits (Cahokia Alluvium) overlie the Mackinaw Member in the floodplain of Killbuck Creek. The surficial deposits south of the Site are mapped as the Esmond Member of the Glasford Formation, a silty clay till (Figure 3-2). The Esmond Member was deposited during the Illinoian Stage of the Pleistocene (Berg, et al., 1984). The Esmond Member has an upper silty facies and a lower silty clay facies. It is gray and contains few pebbles and cobbles. It is characterized by a high illite (clay) content (Willman and Frye, 1970).

3.1.2 Bedrock

The unconsolidated sediments in the region are underlain unconformably by rocks of the Ordovician, Cambrian, and Precambrian Systems (Figure 3-2). The topography of the bedrock surface in the vicinity of the site is shown in Figure 3-3. In Illinois, the Wisconsin Arch has gentle slopes (20 ft per mile), leading to the presence of the Galena Group at the bedrock surface over a broad area (Berg, et al., 1984). The Galena Group of the Ordovician System dominate the bedrock surface in the region. Older rocks of the Platteville and Ancell Groups are at the bedrock surface, as shown in Figure 3-4, due to erosion which formed the the preglacial Rock, Troy, Pecantonica, and Sugar Bedrock Valleys. The WRL Site appears to be on the eastern edge of the preglacial Rock River Valley.

The Galena and Platteville Groups are primarily composed of carbonate rocks (90%), and are subdivided based on the amount of silt and clay. Disseminated siliceous minerals, shale beds, chert nodules, thin sandstone beds, and thin bentonite beds make up the remainder of these two groups. Other features which are used to identify the strata are fossils, chert content, bedding, shale partings, corrosion surfaces, and calcarenites. These variations affect the physical appearance of the strata. The argillaceous (clayey) units trend toward being finer grained, less dolomitic, thinner bedded and weather to a smoother vertical face, whereas the more dolomitic strata, with lesser amounts of fine grained materials, are vesicular and vuggy (Willman and Kolata, 1978).

The Galena Group is 250 to 275 ft thick where overlain by the Maquoketa Shale Group (Willman and Kolata, 1978) (Figure 3-5). The thickness of the Galena is significantly less (due to erosion) where the Maquoketa Shale is absent. In Winnebago County, the Maquoketa Group is present in the southeast portions of the county and in isolated areas of the northeast portion of the county (Berg, et al., 1984), but is not present in the immediate vicinity of the Site. The high purity carbonate Kimmswick Subgroup dominates the Galena Group in thickness (Willman and Kolata, 1978). The Galena Group is generally medium to coarse grained dolomite, while the Platteville is finer grained and thinner bedded (Berg, et al., 1984). The Platteville Group is about 45 ft thick in northwestern Illinois and thickens to the east, reaching 115 ft in thickness near Rockford (Willman and Kolata, 1978). The elevation of the base of the Platteville in the vicinity of the WRL Site is approximately 500 ft MSL, as shown in Figure 3-6.

The Galena and Platteville Groups are underlain by the Ancell Group (Ordovician) which consists of two formations; the Glenwood Formation and the St. Peter Formation (Figure 3-5). The Glenwood Formation is comprised of interbedded dolomite, sandstone and shale. The St. Peter is a fine to coarse grained sandstone. The St. Peter is the surficial bedrock unit at the base of the preglacial Rock, Pecatonica, Sugar, and part of the Troy bedrock valleys (Berg, et al., 1984).

Below the Ancell Group is the Cambrian System, which consists of Potosi, Franconia, Ironton-Galesville, Eau Claire, and Mt. Simon Formations (Figure 3-5). These formations consist of sandstone, dolomite, and shale. The Cambrian System is underlain by Precambrian granite (Berg, et al., 1984).

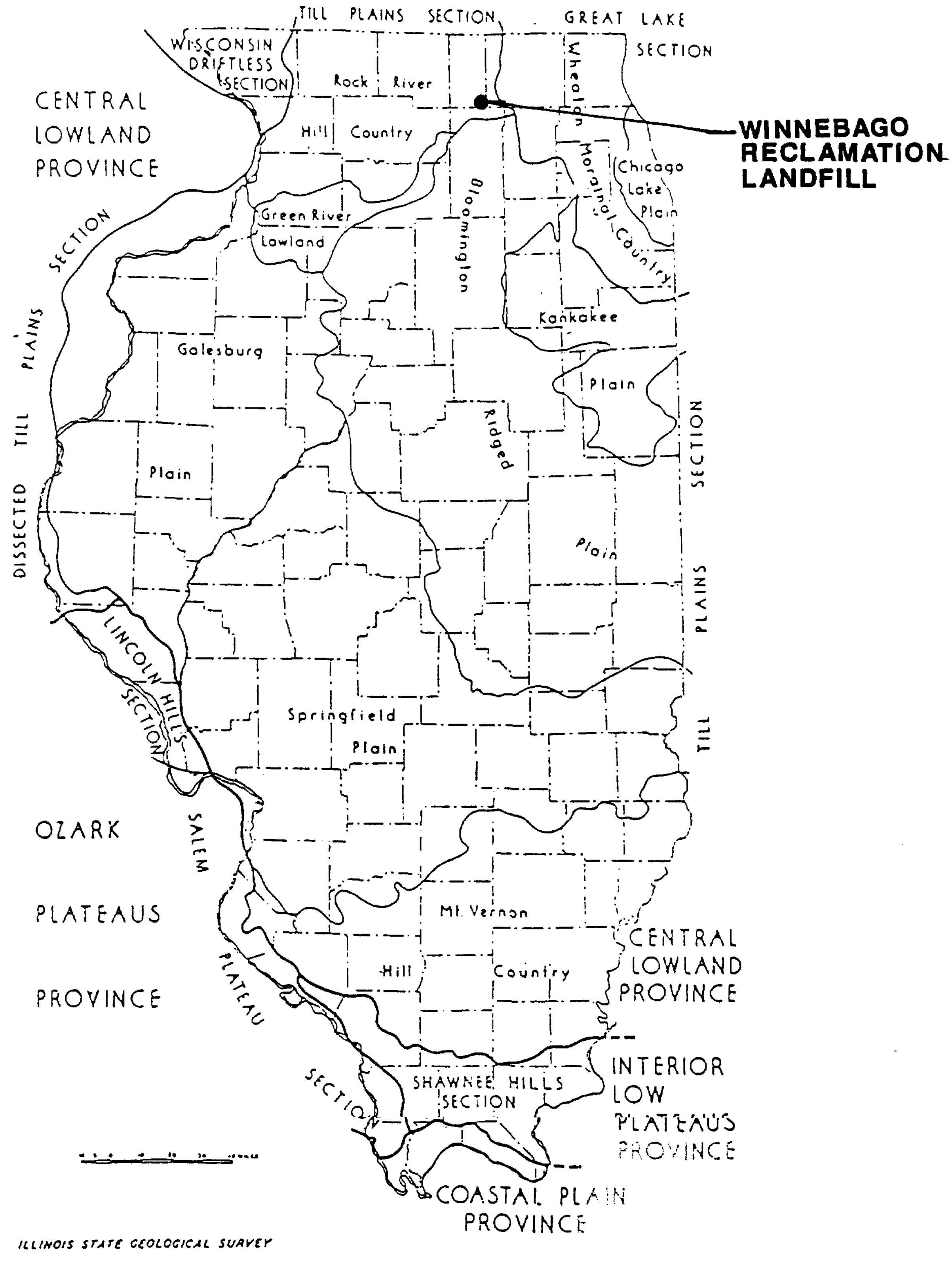
3.2 Regional Hydrogeology

The aquifers in northern Illinois can be divided into two major groups; glacial drift aquifers and bedrock aquifers. The glacial drift aquifers are largely sand and gravel outwash deposits filling the preglacial bedrock valleys. These coarse-textured deposits are typically more than 100 ft thick in the Rock River Valley, with thicknesses in excess of 300 ft being noted. The Silurian and Ordovician dolomites are the major upper bedrock aquifers in northern Illinois (Hackett, 1960).

The Galena-Platteville Groups can be considered one hydrostratigraphic unit. The permeability of these Groups is probably the result of a well developed joint and fracture system, since wells penetrating the dolomites consistently yield water (Hackett, 1960). Although the dolomite has primary porosity, flow is mainly a function of secondary permeability through joints, fractures, and solution openings (Kay, et al., 1989). Hackett (1960) suggests solution activity has not progressed to the point of cave formation. The regional direction of groundwater movement is toward the Rock River from both the northeast and northwest. Groundwater is discharged to the river or to the outwash deposits beneath the river. The water table in upland areas commonly is below the bedrock surface (Hackett, 1960).

According to White (1969), carbonate aquifers can be classified into diffuse-flow and free-flow aquifers. Free-flow aquifers are characterized by well-integrated systems of solution modified conduits. Diffuse-flow aquifers are characterized by limited solution modification of joints, fractures, and bedding planes. On a macro scale, diffuse flow aquifers such as the Galena-Platteville aquifer generally obey Darcy's Law, thus standard aquifer test methods should give reasonable results (White, 1969).

The Ordovician sandstone aquifer (Ancell Group) is hydraulically separated from the overlying Galena-Platteville dolomite aquifer by the low permeability shales and dolomites of the Glenwood Formation, where present. The Glenwood Formation ranges up to 60 ft thick in southeastern Winnebago County. Within the Ancell Group, groundwater generally occurs under artesian or confined conditions, but limited areas of the Ordovician sandstone aquifer are under water table conditions, such as those areas where the rocks are covered by glacial drift. In the Rockford area, the sandstones generally discharge to the Rock River, except where pumping from municipal wells has induced a cone of depression, modifying flow direction (Hackett, 1960).



MAP FROM LEIGHTON, et.al.,1948

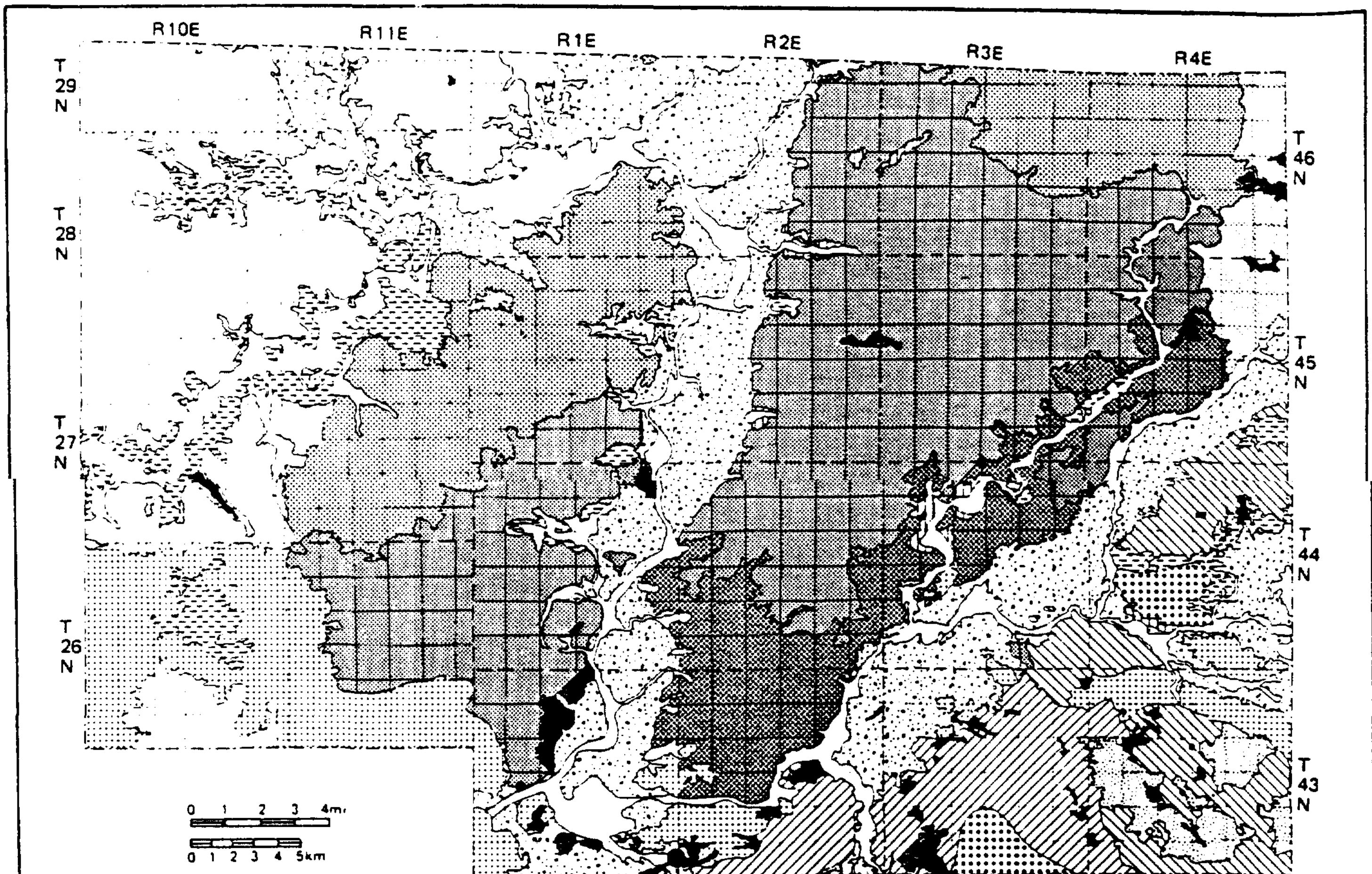
north

FIGURE 3-1

WARZYN	PHYSIOGRAPHIC DIVISIONS OF ILLINOIS
	WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS

DWN D.L.L. | APP'D MAH | DATE 2/10/89 | 13160 | A2

TELEDYNE POST N67178



WINNEBAGO RECLAMATION LANDFILL

FORMATION Member	
CAHOKIA ALLUVIUM	(c) sand, silt and clay deposited by modern rivers and streams
GRAYSLAKE PEAT	(gl) peat and muck often interbedded with silt and clay
EQUALITY	
Carmi	(ec) silt and clay deposited in glacial lakes
Dolton	(ed) medium-to-coarse sand depošted in former glacial lakes
HENRY	
Mackinaw	(hm) thick deposits of sand and gravel in major river valleys
Wasco	(hw) near ice deposits of poorly sorted sand and gravel
WINNEBAGO	
Capron	(wic) pinkish-brown friable silt loam till
Clinton	(wicl) yellowish-tan, very friable, very sandy till
Argyle	(wia) pinkish- or buff-tan often friable sandy till
Nimtz	(win) gray-brown or buff often compact sandy or sandy loam till
GLASFORD	
Unnamed outwash	(g-o) sand and gravel deposit overlying Belvidere Till
Belvidere	(gbl) pinkish or pale brown silty or silt-loam till
Esmond	(ge) grayish-brown silty clay till
Oregon	(gor) pinkish-brown or buff-tan sandy till
Creston	(gc) pinkish-brown silty clay loam till
Ogle	(go) yellowish-brown or buff sandy loam till

FROM BERG et. al. 1984

FIGURE 3-2

north

WARZYN ENGINEERING INC

MAP OF SURFICIAL DEPOSITS IN BOONE & WINNEBAGO COUNTIES

WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS

DWN D.L.L. APP'D [illegible] DATE 3/10/89 13160 A3

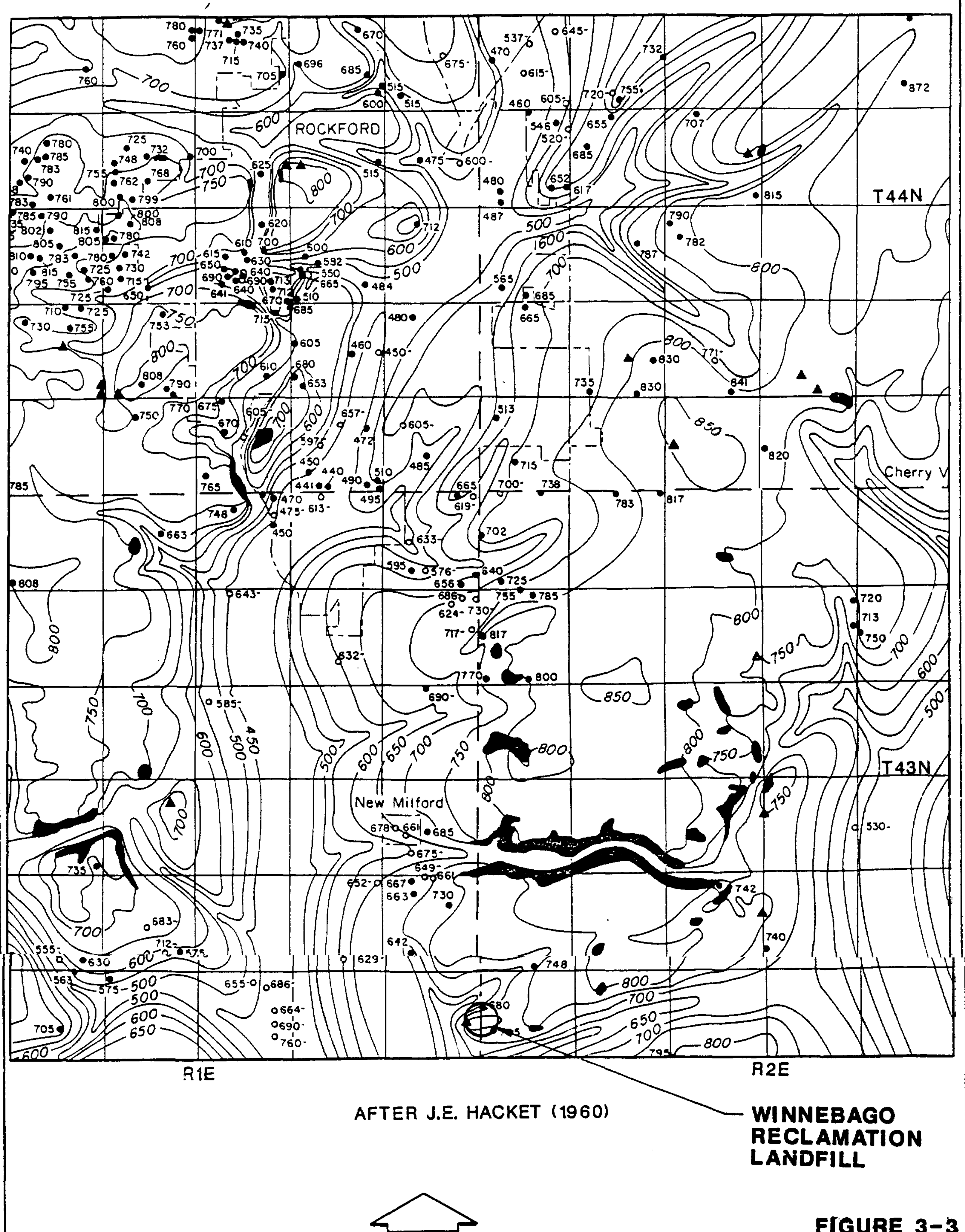
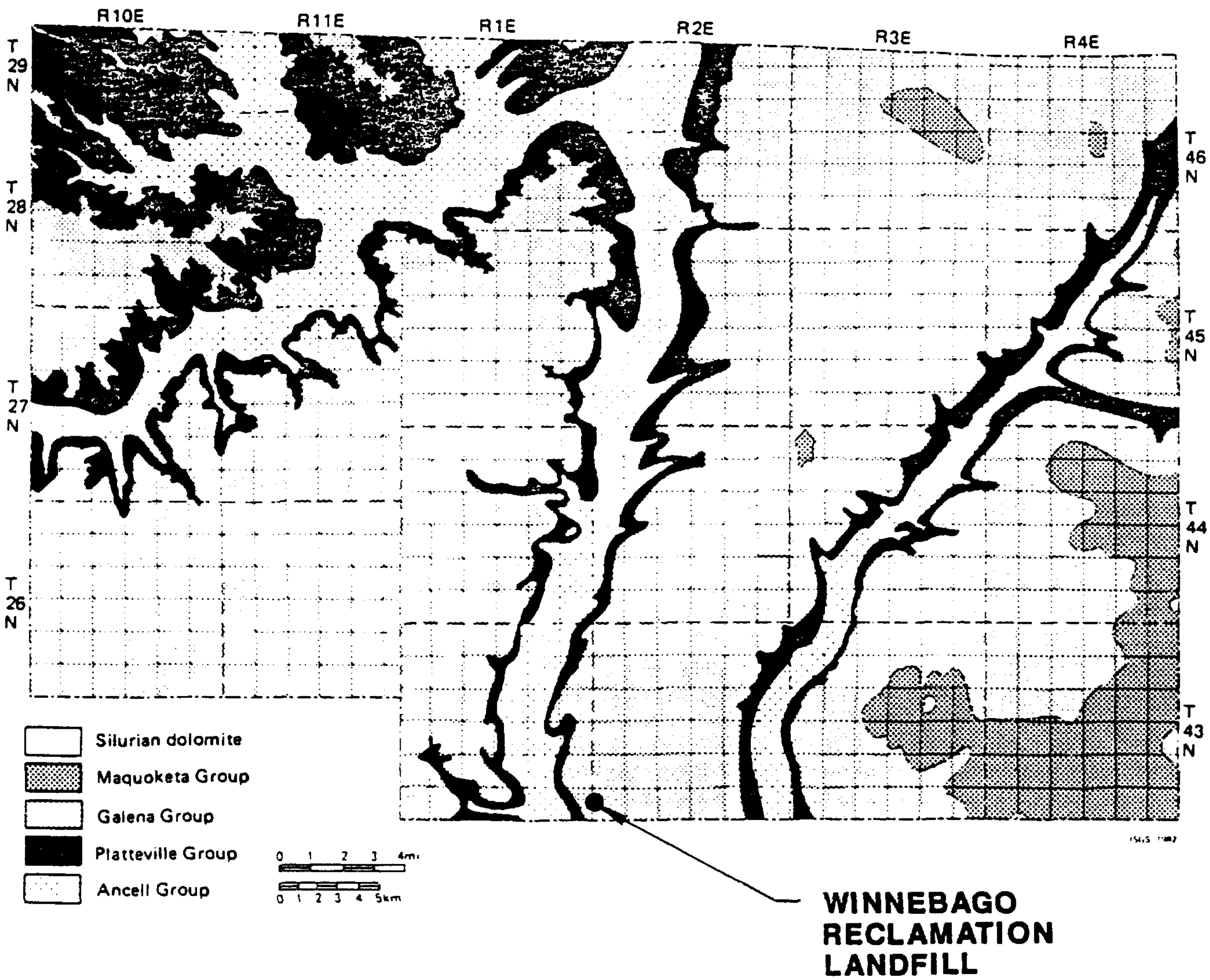


FIGURE 3-3

WARZYN ENGINEERING INC	TOPOGRAPHY OF BEDROCK SURFACE
	WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS

DWN D.LL | APP'D [illegible] | DATE 2/10/84 | 13160 | A4

TELEDYNE POST N67178



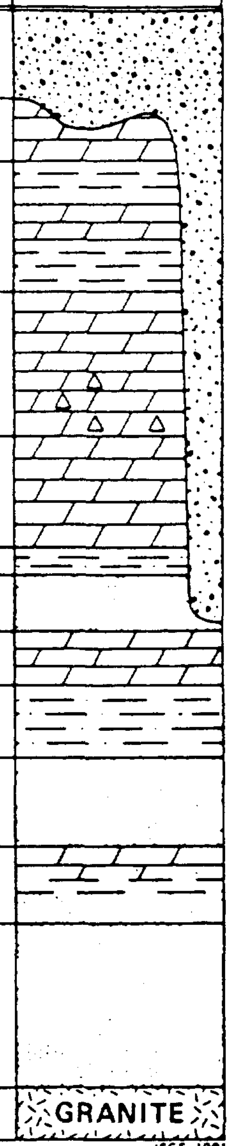
AFTER BERG et. al. 1984

FIGURE 3-4

DWN D.C.L	APP'D	DATE 2/10/84	13160	A5	WARZYN ENGINEERING INC	AREAL GEOLOGY OF THE BEDROCK SURFACE WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS

TELEDYNE POST N47178

SYSTEM	GROUP	FORMATION & THICKNESS	GRAPHIC COLUMN
QUATERNARY 0 0.7 m.y. B.P.		0 - 450 ft	
SILUR. 405 440 m.y. B.P.		50 ft	
ORDOVICIAN 440 490 m.y. B.P.	Maquoketa	150 - 200 ft	
	Galena	250 ft	
	Platteville	100 ft	
		Glenwood 5 - 60 ft	
	Ancell	St. Peter 200 - 400 ft	
CAMBRIAN 500 515 m.y. B.P.		Potosi 50 - 100 ft	
		Franconia 50 - 100 ft	
		Ironton - Galesville 75 - 170 ft	
		Eau Claire 350 - 450 ft	
		Mt. Simon 1000 - 1600 ft	
PRECAMBRIAN			GRANITE



FROM BERG et. al. 1984

FIGURE 3-5

DWN D.LL	APP'D	DATE 2/11/89	13160	A6	WARZYN ENGINEERING INC	STRATIGRAPHIC COLUMN FOR BOONE & WINNEBAGO COUNTIES WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS

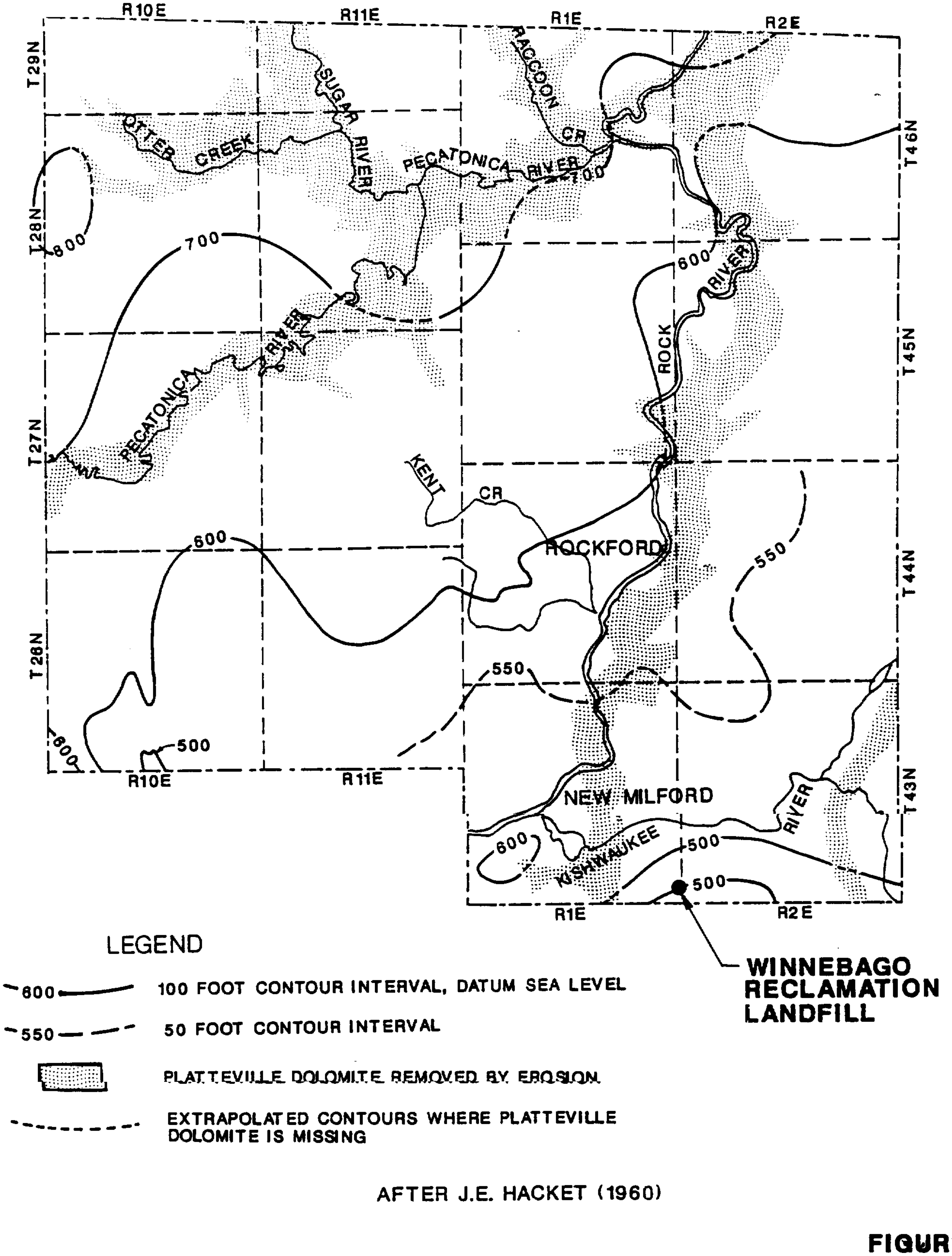


FIGURE 3-6

ELEVATION CONTOURS OF THE BASE OF THE PLATTEVILLE DOLOMITE IN WINNEBAGO COUNTY

WARZYN ENGINEERING INC

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

DWN D.L.L. | APP'D [illegible] | DATE 2/10/89 | 13160 | A7

SECTION 4
RESULTS AND DISCUSSION

Findings of the RI are discussed by medium in the following subsections. The evaluation of the findings leads to an understanding of the physical features influencing the movement of contaminants, particularly in the groundwater; identification of the chemicals present and of potential concern in the various media at the Site; the media potentially impacted; and the identification, impact, or potential influence of other potential sources.

4.1 Leachate

4.1.1 Volatiles

The 12 leachate samples analyzed by GC/MS provided acceptable data and therefore were used for leachate characterization. The samples locations are shown on Figure 2-1. The GC data were generally considered estimated or unusable in large part due to matrix interferences. The leachate samples analyzed by GC/MS., generally contained aromatic VOCs such as benzene, ethylbenzene, toluene, and xylenes at higher concentrations than chlorinated compounds such as PCE and TCE (Tables 4-1 and 4-2). Benzene was detected six times ranging from 2J ug/L to 6.2B ug/L. Ethylbenzene was detected in 11 samples, with concentrations ranging from 5 ug/L to 77J ug/L, averaging 45 ug/L. (The J is a data qualifier denoting an estimated value, B denotes constituent also found in laboratory blank). Toluene was detected in all the samples analyzed by GC/MS and ranged in concentration from 18J ug/L to 380JD ug/L, averaging 149 ug/L. (D is a laboratory qualifier indicating the sample was diluted to get the concentration with in the range of the instrument.) Xylenes were also found in all the samples analyzed by GC/MS, ranging in concentration from 69 ug/L to 300J ug/L, averaging 151 ug/L. Tetrachloroethene was detected in only two samples at concentrations of 17 ug/L (LLG4) and 0.7 ug/L (LLD1). Trichloroethene was detected in two of the samples at 1J ug/L (LLD1) and 29 (LLE3). Total 1,2-dichloroethene was detected in nine samples, generally ranging in concentration from 4.3 ug/L to 24 ug/L with one sample reaching 2,100 ug/L (LLE3), with an overall average of 268 ug/L. Vinyl chloride was also detected in nine samples and the concentrations ranged from 2J ug/L to 270D ug/L, while averaging 53 ug/L.

The Jordan (1984) leachate sample results generally fell within the same concentrations (Table 4-1). The comparison of the Jordan (1984) data to the RI data shows that the current leachate VOC composition is similar to the VOC makeup of leachate in 1984, indicating that the leachate has not changed significantly over this time span.

4.1.2 Semi-Volatiles

Semi-volatiles were targeted for analysis during Rounds 1 and 5. Target list compounds were generally detected in fewer than four of the 12 samples, but 2,4-Dimethylphenol (33-160 ug/L), naphthalene (6-50 ug/L), and bis (2-ethylhexyl) phthalate (80-1200 ug/L) were all reported as detected in six of 12 samples. The results are summarized in Table 4-3 (see Appendix C for further details).

Analytical protocols require the laboratory to attempt to detect up to 20 compounds of greatest apparent concentration and provide tentative identifications. For each of the 12 samples analyzed, 20 such compounds were reported, however, most compounds were not identified.

4.1.3 Pesticides/PCBs

The PCB Aroclor-1242 was detected in two of the Round 1 leachate samples at "estimated" concentrations of 6.9J ug/L and 3.3J ug/L. No other pesticides/PCBs compounds were reported as detected in Round 1 leachate samples, however, the presence of Aroclor-1242 created some uncertainty about results for other PCBs and pesticides that may co-elute (i.e., reach the detector over the same time interval) with Aroclor-1242. Leachate samples were again tested in Round 5 to confirm the Round 1 results. Aroclor-1242 was detected at five of the six Round 5 sampling locations at concentrations ranging from 2.7 to 3.7 ug/L.

Round 5 leachate also showed Aroclor-1248, Aroclor-1254, and Aroclor-1260 at concentrations ranging from 1.8-3.8 ug/L; and Aldrin, Heptachlor Epoxide, Endosulfan Sulfate, Gamma-Chlordane, Endrin, Alpha-BHC, Beta-BHC, Delta-BHC, and Gamma-BHC (Lindane), each at concentrations less than 1 ug/L (see Appendix C for further details).

4.1.4 Inorganics and Indicator Parameters

Results of most inorganics analyses are summarized in Table 4-4 (see Appendix C for further details) and compared with typical values for sanitary landfill leachate by Cope, et al., (1983). Although Cope, et al., (1983) acknowledges considerable variability in leachate composition among landfills, their results are considered useful for comparison purposes. In general, sanitary landfill leachates have a high inorganic component. For the most part, results of WRL Site analyses are within typical ranges. However, WRL Site leachate results do indicate higher than typical sodium and chloride concentrations. Chloride ranged from 1,160 mg/L to 17,300 mg/L, and sodium ranged from 968 to 10,200 mg/L. Potassium ranged from 608 to 1,750 mg/L. Calcium and magnesium ranged from 29.9 to 241 mg/L and 30.8 to 812 mg/L, respectively. The iron concentration ranged from 5.47 to 263 mg/L.

4.2 Area Geology

4.2.1 Unconsolidated Deposits

The unconsolidated materials are predominantly sand and gravel deposits near the WRL Site. Generally, a sequence of sand and gravel or fine to coarse sand occurs in the lower portions of the unit, changing to silt or clay near the ground surface. The uppermost silt and clay, where present, is generally 1 to 3 ft thick, but was found to be as much as 8 ft thick (G109; Warzyn, 1985 and G114) on the south side of the WRL Site. Fine-grained soil was present south of the Acme Solvent Site at a thickness of at least 40 ft (B3; E&E, 1983) (see below). Portions of the basal sand and gravel were sometimes recognized as weathered bedrock (B7, B12, B16A; E&E, 1983) (G111; Warzyn, 1985). These severely weathered zones ranged from 6 to 15 ft thick. To the west, the unconsolidated sediments thicken, as they fill the bedrock valley. The soil types are predominantly fine to coarse sands with occasional fine to coarse gravel zones 11 ft (G118) to 40 ft (MW106; Jordan, 1984) thick (Drawing 13160-F2).

The fine grained deposits to the south of the WRL Site are predominantly silts and clays with occasional thin sand lenses (B14, B3, B8; E&E, 1983) (G111; Warzyn, 1985). These deposits appear to be part of the Esmond Till Member. Boring G111 indicates that the silts and clays are up to 24 ft thick and rest on 5 ft of fine to coarse sand with some gravel underlain by weathered bedrock (gravel with some sand). Boring B3 indicates the silt and clay, interstratified with sand, are at least 40 ft thick.

The thickness of the unconsolidated materials range from 8 ft at the Acme Solvent Site (B4; E&E, 1983) to greater than 70 ft (P4R) at the western boundary of the WRL Site (although bedrock is exposed in places at the Acme Solvent Site (Jordan, 1984). The body of the unconsolidated deposits thicken from east to west, forming a relatively thin, but variable mantle over the bedrock upland in the east, and filling the deep bedrock valley to the west (see Drawing 13160-F2). The transition from a bedrock aquifer to an unconsolidated sand and gravel aquifer begins beneath the eastern margin of the landfill where the bedrock surface slopes downward forming the preglacial bedrock valley wall. Under the western portion of the WRL Site, bedrock was encountered in P5 (now abandoned; Jordan, 1984) at 671 ft MSL, indicating unconsolidated deposits greater than 50 ft thick (Jordan, 1984) (see Figure 4-1 for location). The full thickness of valley-fill sediments west of the Site was not penetrated in this study, but based on regional information, the thickness is expected to be about 100 ft under Killbuck Creek near the WRL Site. The regional bedrock surface topography map shows the elevation of the bedrock surface to be approximately 600 ft MSL under Killbuck Creek at the western margin of the WRL Site (see Figure 3-3).

4.2.2 Bedrock

The bedrock surface elevation is highly variable due to paleoerosional features. The bedrock map, Figure 4-1, is based on available data. The actual bedrock surface is expected to show more variability (logs for borings SB-1 to SB-5 are included in Appendix F; see Figure 4-1 for locations). To the east of the WRL Site is a bedrock upland. A bedrock valley begins to form under the Site and deepens and widens to the west. The regional bedrock surface topography map illustrates the bedrock upland as a northeast-southwest trending ridge with valleys to the north and south (Figure 3-3). The western ends of these valleys terminate in a north-south trending valley. The north-south trending valley slopes downward to the south. The full thickness of the Galena-Platteville Groups (i.e., dolomite aquifer) was not penetrated in this study. Regional information suggests it ranges from 250 ft thick in the bedrock upland (i.e., the base of the Platteville is at approximately 500 ft MSL) to less than 100 ft thick in the adjacent bedrock valley to the west.

The bedrock near the WRL Site is composed of dolomite, with chert layers or nodules commonly noted throughout the dolomite. Shale partings and coatings were noted only below 695 ft MSL. The dolomite is generally fractured throughout the interval sampled. The fractures are dominantly horizontal bedding planes, frequently cross-cut by high angle or vertical fractures. Vugs (void spaces) are consistently found throughout the dolomite, with their frequency ranging from slightly vuggy to very vuggy. Cavernous zones were not noted.

The rock quality designator (RQD) is a ratio of the total length of recovered pieces of rock core greater than 4 in. long to the total length of recovered rock core, reported as a percent. The RQD is a quantitative measure of the degree of fracturing in the sample. A high RQD indicates more competent rock (less fractured), while a low RQD indicates more fractured rock. The RQD of dolomite core samples range widely, from 0 to 100%, averaging 52.5%, with a standard deviation of 28.9% (Appendix A, Warzyn 1985, and E&E, 1983). The wide variability in the RQD provides an indication of the variably fractured nature of dolomite.

A zone of up to 37 ft thick of highly fractured, soft dolomite in the near surface bedrock was encountered during drilling in previous investigations at 752 ft to 715 ft MSL in boring B16A (Warzyn, 1985), at 740 ft to 715 ft MSL in boring B7 (E&E, 1983) and at 737 ft to 732 ft MSL in boring B6D (E&E, 1983). The borings where the fractured zones were encountered are located in the vicinity of the northern intermittent stream, with the RQDs ranging from too soft to core to 28%.

Highly fractured zones (low RQD) were also found between intervals of rock containing few fractures (high RQD), indicating rock competence did not generally improve with depth (e.g., B12 (E&E, 1983), G109A (Warzyn, 1985), G113A, B11A (Warzyn, 1985), and B6D (E&E, 1983)). One zone of low RQD between zones of higher RQD was found near the interval 690 ft to 710 ft MSL in borings B6D (E&E, 1983), B16A (Warzyn, 1985), and G109A (Warzyn, 1985). Within this interval, an intensely fractured to crushed zone was noted in the log for STI-5D, where the cuttings changed from wet to saturated (STI Report, 1990). The geophysical logs (density and neutron) for boring STI-5D also indicate a low density (high porosity) zone (695 ft - 707 ft MSL) with a higher water content (STI Report, 1990). Since the fractures in the dolomite aquifer are reportedly dominantly horizontal in orientation and fracture zones were noted in the above described borings at about the same elevation range (690 ft - 710 ft MSL), it is possible that the same fracture zone was encountered at each location.

4.3 Area Hydrogeology

The uppermost aquifer encountered in the vicinity of the WRL Site changes in character due to the abrupt slope of the bedrock surface beneath the Site (Drawing 13160-F2). East of the WRL Site, and below approximately its eastern one quarter, the water table occurs within the fractured dolomite bedrock. An approximately northeast-southwest line drawn from well B10 to well B13 delineates the boundary of the occurrence of the water table in bedrock to the east and the unconsolidated materials to the west. Regardless of the type of matrix material of the aquifer, the uppermost saturated unit in the immediate vicinity of the WRL Site is under water table conditions. South of the Site, the water table occurs in the silty clay till. The sand and gravel-dolomite aquifer underlying this till is under semi-confined conditions (e.g., see well G111 where water level is in the clay above the top of the aquifer).

Inspection of the potentiometric maps (Figures 4-2 to 4-13) indicates that groundwater flows generally from the bedrock upland (recharge area) east of the WRL Site west to the Killbuck Creek Valley (potential discharge area). Where the water table occurs in the unconsolidated materials in the vicinity of the western portions of the WRL Site, groundwater flow is generally towards Killbuck Creek. Shallow groundwater discharges to Killbuck Creek, but deeper groundwater flows west beneath the creek (Figures 4-10 to 4-13).

The groundwater flow system is divided into three general areas for discussion below:

- Flow beneath the WRL Site
- Flow near Killbuck Creek
- Flow in the bedrock upland

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- Flow near Killbuck Creek
- Flow in the bedrock upland

4.3.1 Flow Beneath the WRL Site

Groundwater flow beneath the Site in the unconsolidated deposits is generally westward towards Killbuck Creek. Beneath the northern portions of the Site, groundwater flow is towards the northwest, while beneath the southern portions of the Site, groundwater flow is towards the southwest. North of the Site, groundwater flow appears to be west to southwest towards Killbuck Creek. South of the Site, groundwater flow appears to be west to southwest towards Killbuck Creek. The hydraulic conductivity of the unconsolidated deposits ranges from 3.7×10^{-1} to 8×10^{-4} cm/s with a geometric mean of 7×10^{-3} cm/s.

Review of the hydrograph on Figure 4-16 shows the water level elevations of wells G115, P3R, B13, G114, and B14, located on the southern half of the WRL Site, follow a generally consistent pattern of smooth declining or rising limbs with few anomalies. Well G115 appears to have water level elevations which are sometimes greater than expected, indicating groundwater in the vicinity of this well is receiving excess recharge. The anomalies were noted only in the springtime when precipitation and snowmelt are the highest, indicating a relationship to local precipitation infiltration. The observed chloride concentrations in samples from well G115 have increased in Rounds 3 and 4, indicating the presence of WRL leachate (also see Inorganic/Indicator discussion in Section 4.4.1), which may be contributing to the water level anomaly.

4.3.2 Flow Near Killbuck Creek

Shallow groundwater appears to discharge to Killbuck Creek (Figure 4-11 to 4-13). The staff gauge water level measurements in Killbuck Creek are lower than water level elevation measurements from nearby wells, indicating a gradient towards the creek. The water level measurements at well nest P3R/P4R indicate the potential for upward vertical gradients at this location, indicating the potential for groundwater to discharge to Killbuck Creek.

The potential for flow beneath the creek is indicated by the water level measurements for wells P1/MW106 which generally show no measurable difference in elevations, indicating generally horizontal flow. Three measurements at nest P1/MW106 indicated potential sometimes for downward vertical gradients. (One of these three measurements

(11/9/88), however, is reported to be in error. According to Mr. B. Kay of the USGS the depth to water measurement was made from the outer casing instead of the inner casing.) (April 13, 1989 letter from Mr. B. Schorle, U.S. EPA to Mr. J. Hill, Warzyn). The water level elevations for the shallow well G116 are generally about the same, but sometimes less than those in the deeper well G116A. This indicates that upward vertical gradients may sometimes be present, but groundwater flow in generally horizontal at these wells screened in sand and gravel west of Killbuck Creek.

4.3.3 Flow in the Bedrock Upland

Groundwater flow in the bedrock upland is generally towards the west. There are two features which affect the generally westward flow direction; the presence of a high permeability zone between the Acme Solvent Site and the southeast corner of the WRL Site, and groundwater mounding near the unnamed stream adjacent to the Acme Solvent Site. The effect of a groundwater mound is a local gradient reversal (i.e., radial flow locally). The effect of the high permeability zone is high gradients and converging flow at the upgradient end of the zone, low gradients within the zone, and high gradients and diverging flow at the downgradient end of the zone.

4.3.3.1 Mounding in the Bedrock Upland

West is the typical downgradient direction, but an eastward gradient was observed (i.e., a groundwater mound) on April 6, 1988 (Figure 4-2) in the upland bedrock area east of the WRL Site. The April 6, 1988 water level elevation at well B7 is greater than at any other well to the east of the WRL Site on that date (Figure 4-15). It is thought the mounding is due to higher localized recharge rates in this area from the north unnamed intermittent stream. The presence of groundwater mounds were again noted in the upland bedrock area east of the WRL Site on February 5, 1990 and April 20, 1990 (Figures 4-11 and 4-13). The February 5, 1990 water level elevations at wells B7 were greater than any well to the east of the WRL Site on that date, as shown on the hydrograph in Figure 4-3. Similarly, on April 20, 1990, the water level elevations at wells B4 and B7 were again greater than any well to the east on that date (Figure 4-15).

The presence of groundwater mounds was also noted in previous investigations related to recharge from the north unnamed intermittent stream. Water table maps prepared from water level data collected in 1985 and 1986 depicted a groundwater mound centered around wells B4 and B7 (Appendix F), associated with the unnamed intermittent stream north of the WRL Site. In March 1985, a marked drop in flow relative to upstream was observed in the north unnamed intermittent stream near well B16 during heavy precipitation and high surface water runoff, indicating localized recharge to groundwater from the unnamed intermittent stream was occurring. At a culvert in the unnamed intermittent stream upstream (east) of well nest B16/B16A, flow was estimated to be 133 cubic feet per second (59,700 gallons per minute). (See Drawing 13160-F1 for culvert location.) A marked decrease in stream flow was noted downstream of the culvert (near well nest B16/B16A), indicating significant surface water loss to the groundwater (Appendix F, July 12, 1985 report).

4.3.3.2 High Permeability Zone in the Bedrock Upland

In the bedrock upland, fractured zones sandwiched between more competent zones suggests that a high permeability zone exists. The presence of this high permeability zone could create a narrow zone of preferential groundwater and contaminant transport between the vicinity of the Acme Solvent Site and the southeast corner of the WRL Site. Observations indicating a high permeability zone within the dolomite aquifer are:

- Geologic data obtained during coring
- Geophysical data from borehole logging
- Pumping test results
- Permeability test results
- Water level observations from wells
- Groundwater quality data

The geologic logs for wells B6D, B16A, and G109A indicated the presence of a low RQD (more fractured zone) zone near the elevation interval 690 ft - 710 ft MSL (Drawing 13160-F2) (see Section 4.2.2). The geophysical logging of well STI-5D also indicated the presence of a fractured zone near the elevation interval 695 ft - 707 ft MSL (STI Report, 1990)(also see Section 4.2.2). Since the dolomite aquifer is characterized by horizontal fractures, and the highly fractured intervals were found at approximately the same elevation, it is possible that a continuous fracture zone may exist at this elevation.

A pumping test was performed at the Acme Solvent Site during the STI by the consultant to the Acme Solvent PRPs (see Appendix L of the STI Report, 1990). Pumping test results by their very nature provide non-unique solutions, as several combinations of hydraulic boundaries and aquifer characteristics may explain the same set of data. The early part of the pumping test are easily explained by a uniform aquifer of relatively low permeability. However, the later part of the pumping test deviated from this simple model, suggesting the test had affected a different part of the aquifer. The STI report (1990) indicated that a pervasive fracture zone in the dolomite aquifer, among other possibilities, can explain the pump test results. The following explanations were offered by the STI report authors:

- Double-porosity effects of a fractured medium
- Delayed gravity response of the water table
- Recharge effects from the St. Peter Sandstone
- The presence of a hydrologic recharge feature in the Galena-Platteville (dolomite aquifer)

The hydrologic recharge feature would be a more permeable fracture zone.

Rapid recovery of the water level after the cessation of the pumping test was reported in Appendix L of the STI (1990), and the following explanations were offered by the report authors:

- A hydrologic (recharge) boundary may have been present near the well. Such a boundary could result from a pervasive fracture zone of relatively high permeability or a solution feature within the Galena-Platteville (dolomite aquifer). Alternatively, a source of recharge could be related to the presence of the St. Peter Sandstone, a permeable hydrostratigraphic unit below the Galena-Platteville-Glenwood sequence.

- Hydraulic responses may have been affected by delayed yield mechanisms. If this was caused by delayed gravity response, dewatering at the water table would have provided a source of external recharge to the system. Delayed yield could also have been related to double-porosity behavior between fractures and rock matrix. In this case, depressurization of the rock matrix would have provided a source of recharge.

In both explanations offered by the authors of the STI report (1990), the presence of a pervasive, highly fractured zone could explain the observed water level responses.

The permeability test results document the wide range of permeabilities characterizing the dolomite aquifer, as would be expected given the variably fractured nature of the aquifer (Table 2-1). The hydraulic conductivity of the dolomite bedrock ranges from 2.4 x 10^{-2} to 5.4x10^{-6} centimeters/second (cm/s) with a geometric mean of 2.3 x 10^{-4} cm/s. While none of the wells listed in Table 2-1 are located within the more fractured zone discussed above, the permeability test results indicate the measured permeability for the dolomite bedrock aquifer ranges over three orders of magnitude from high to very low permeability.

There are three related features shown on the water table maps (Figures 4-2 to 4-13) that indicate a high hydraulic permeability zone is present between the Acme Solvent Site and the WRL Site. These features are:

- A zone of steep hydraulic gradients with the potentiometric groundwater contours convex in the upgradient direction in the vicinity of well B6S, indicating converging flow lines

- A zone of very low gradient located between wells B6S and G109, indicating groundwater flow with very low hydraulic gradient (head loss)

- A zone of steep hydraulic gradients with the potentiometric contours convex in the downgradient direction near the southeast corner of the WRL Site indicating diverging flow away from the area

Examples of hydraulic gradients were calculated for each of these zones noted on the potentiometric maps. At the east and west ends of the high permeability zone, the hydraulic gradients were 0.016 ft/ft and 0.011 ft/ft respectively, but within the high permeability zone the hydraulic gradient was much lower at 0.002 ft/ft (Figure 4-7).

The effect of a high permeability zone on groundwater flow directions is quantitatively illustrated in Figure 4-17. This figure is the result of a numerical model showing groundwater head and flow lines in the vicinity of a high permeability zone. A high permeability zone situated in a lower permeability area using a permeability difference of 100 times. Note the concave inward potentiometric contour lines showing converging inward flow at the upgradient end of the high permeability zone and diverging outward flow at the downgradient end of the high permeability zone. Due to the converging flow at the upgradient end of the high permeability zone, the gradient outside of the high permeability zone is greater than elsewhere in the aquifer. There is relatively little head loss through the high permeability zone due to its ability to easily transmit water. At the end of the high permeability zone, the gradient is once again higher than elsewhere in the aquifer due to the diverging flow.

For comparison purposes, a zone of lower permeability (100 times less) within a higher conductivity area is shown in Figure 4-18. Note that the effect on the potentiometric surface due to the low permeability zone differs from that of the high permeability zone. The comparison of the two types of zones discussed above in plan view are also valid in cross-section. Winter (1976) provides an excellent discussion of the effects of the presence of high permeability zones at depth on groundwater flow paths using cross-sectional flow nets.

Considering the high permeability zone hypothetical groundwater flow model in three dimensions, the groundwater flow system has a funnel-shape on the upgradient end converging flow into the high permeability zone. On the downgradient end, the funnel shape is reversed so that flow diverges out of the high permeability zone. Contaminants entering the groundwater flow system at the upgradient end of the high permeability zone would be "funneled" into a relatively narrow zone within the high permeability zone. At the downgradient end of the high permeability zone, the contaminants would "exit the wide end of the funnel". This model is consistent with the contaminant distributions observed in the fractured dolomite aquifer, where contaminants have not been found in high concentrations between the two Sites in wells screened above or below the fractured zone. At the downgradient end of the fracture zone, contaminants are widely dispersed in the aquifer (See Section 4.4).

Decreasing water levels to the north, south, and west in the vicinity of the eastern end of the WRL Site are consistent with the diverging flow discussed above. Water level elevations from well G112, which is east of Lindenwood Road, have been consistently higher than other wells in the immediate vicinity (including G109, B12 and G113), indicating that groundwater flow is toward the landfill at this point (Figure 4-14). Well B12, located between wells G109 and G113, approximately 60 ft south of G109 and about 100 ft northeast of G113, has exhibited water level elevations which are generally greater than those for either G109 or G113, indicating that a groundwater flow divide is located in the vicinity of well B12 (Figure 4-14). Groundwater flow is towards the landfill in the vicinity of well B12. On two of the 12 occasions during the April 1988 and April 1990 investigation period, water levels indicated no measurable differences between G109 and B12 (May 17, 1988 and June 16, 1988). On one occasion, the water level measured at G109 (718.21 ft MSL) was slightly greater than the water level measured at well B12 (718.10 ft MSL).

Naturally occurring downward gradients typically are indicative of recharge areas while upward gradients indicate discharge areas. Variations from these simple rules indicate the potential for heterogeneous conditions within the aquifer. Heterogeneous means aquifer properties such as permeability may vary both horizontally and vertically. For example, upward gradients in an area where recharge conditions (downward gradients) are expected, indicate the potential presence of heterogeneous conditions within the aquifer. The upward gradients may be explained by the presence of a higher permeability zone located above the deeper measuring point (i.e., screened interval of well) causing upward groundwater to flow to the higher permeability zone. Downward gradients towards the high permeability zone also are present in this situation (i.e., flow is from above and below the high permeability zone).

Figure 4-19 is an east-west cross-section with potentiometric contours prepared for the October 25, 1988 water level data, with no corrections for vertical exaggeration following Winter (1976). The figure depicts a strong downward component of groundwater flow in the vicinity of well B4. In the vicinity of well nest B6S/MW105/B6D, the potentiometric lines are concave inward (west) indicating groundwater flow converges inward from both above well B6S and below well MW105. In the vicinity of well nest STI-5S/STI-5I/STI-5D, the potentiometric contour line is vertical, indicating horizontal groundwater flow. Near well nest G109/G109A, the potentiometric contour lines are concave inward towards the east (i.e., divergent outward), indicating upward and downward groundwater flow components as well as a horizontal one. This configuration is similar to that simulated on the downgradient end of the high permeability zone (see Figure 4-17).

4.4 Groundwater Chemistry

4.4.1 Inorganics/Indicators

As discussed earlier in Section 4.2, the WRL leachate has a high inorganic component, consistent with other solid waste landfill leachates. The inorganic components of the leachate can be used to discriminate between WRL leachate affected wells and unaffected wells. Figure 4-20 is a trilinear plot of the major cations calcium (Ca), magnesium (Mg), and sodium plus potassium (Na + K) (as percent of total meq/l), which are often used to discriminate between groundwater types. The WRL leachate samples plot as a Na + K rich water, while upgradient or unaffected wells plot as Mg and Ca rich, forming the end points of a continuum encompassing wells apparently affected by the WRL Site. Johnson and Cartwright (1980) used this approach and also found landfill leachate to be Na + K rich and were able to discriminate contaminated versus uncontaminated samples on this basis. The U.S EPA reported that chloride, sodium, and bicarbonate "are frequently the ions of choice for tracing leachate in groundwater" (U.S. EPA, 1980; p.7).

The major cation discrimination of leachate impacted groundwater is supplemented by the strong positive correlation between Na + K and the chloride ion (r^2 = 0.998) (Figure 4-21). The chloride content of the WRL leachate is greater than typical landfill leachate (Table 4-4), and significantly higher than that of groundwater in the area. The near unity correlation between chloride and Na + K, and the large chloride concentration contrast between groundwater and WRL leachate, indicates that chloride can be used to discriminate between WRL leachate affected and unaffected wells.

Johnson and Cartwright (1980) were also able to discriminate between leachate contaminated and uncontaminated water samples on the basis of the higher chloride content of the contaminated samples. Chloride is widely recognized as a conservative, non-reactive parameter in groundwater systems (Bently, et al., 1986; Freyberg, 1986; Roberts, et al. 1986; Wehrmann, 1983; Hem, 1989; Feth, 1981), which means it does not biodegrade or react with the aquifer matrix. The primary attenuation mechanism applicable to chloride is dilution during transport. The conservative non-reactive nature of chloride coupled with the high contrast in chloride concentration between leachate and groundwater make chloride a very good "tracer" of the presence of leachate in groundwater.

Feth (1981), in a review of the chemistry of chloride by the USGS, quoted Piper, et al., (1953) as follows "... only chloride probably remains chemically inert in the zones of contamination. Hence, the amount of this constituent can be taken to indicate the proportion of the mixture." All other components of the leachate should at a minimum exhibit the same level of dilution as for chloride during downgradient transport, but attenuation should be much greater for metals and organic compounds due to their reactive nature. Most metals and organic compounds are reactive and non-conservative in groundwater systems due to processes such as cation exchange, biodegradation, and/or adsorption, which tend to further reduce their concentrations in groundwater, resulting in an attenuation factor greater than that of chloride.

Using chlorides as a "tracer", inspection of Figures 4-22, 4-23, 4-24, and 4-25 show three general areas where chloride concentrations indicate the presence of WRL leachate. These are:

- The northwest quadrant of the WRL Site defined by wells B15R, MW106, P1, P4R, and G116A
- In the vicinity of wells G110 and G114
- In the vicinity of well G115

At the northwest quadrant of the WRL Site, Round 1 groundwater data for wells B15, B15R, MW106, P1, and P4R clearly exhibit elevated chloride concentrations (Figure 4-22). These are the same wells which form the continuum between WRL leachate affected wells (based on Round 1 data) and unaffected wells on Figure 4-20 with respect to the Ca, Mg, and Na + K cations.

In cross-sectional view as shown on Figure 4-27, the WRL Site chloride plume is present only in the upper portion of the saturated zone near well nest B15/B15R/B15P, since the chloride concentration in the Round 1 sample from well B15P (14 mg/l) is low. (See Figure 4-26 for cross-section location.) The WRL Site chloride plume is present deeper in the sand and gravel aquifer downgradient from the WRL Site as indicated by the chloride concentrations observed during Round 1 at well nest P1/MW106 (252 mg/l and 430 mg/l, respectively).

Round 2 groundwater results for chlorides indicate a similar pattern at the northwest quadrant of the WRL Site, except that well G116A (deeper well in nest G116/G116A) showed an increase in chlorides, indicating that the WRL Site leachate plume had migrated west below Killbuck Creek, but had not affected the shallower groundwater as indicated by well G116 (7 mg/l chlorides) (Figures 4-23 and 4-28). Round 2 chloride data indicates the WRL Site chloride plume is again present only in the upper portion of the saturated zone near well nest B15/B15R/B15P, since the chloride concentration in the Round 2 sample from the deeper well B15P (15 mg/l) is again low, consistent with Round 1 data. Downgradient of the WRL Site, both the shallow and deeper portions of the saturated zone east of Killbuck Creek (P1/MW106) show impacts due to WRL leachate similar to Round 1 data.

Round 3 groundwater results for chloride at the northwest quadrant of the WRL Site showed some reductions in chloride concentrations compared to Round 2 data (Figures 4-23 and 4-24). For example, chloride concentrations observed at well P1 decreased from 176 mg/l in Round 2 to 65 mg/l in Round 3, and observed chloride concentrations at well G116A decreased from 99 mg/l in Round 2 to 39 mg/l in Round 3. Similarly, observed chloride concentrations at well P4R decreased from 188 mg/l in Round 2 to 25 mg/l in Round 3. Well P3R exhibited an increase in chloride concentration from 46 mg/l in Round 2 to 72 mg/l Round 3. The chloride concentrations observed at well nest B15/B15R/B15P again indicate the WRL Site chloride plume is present only in the upper portion of the saturated zone, since the chloride concentration in the Round 3 sample from the deeper well B15P (13 mg/l chlorides) was still low. The Round 3 chloride result for well G116 (7 mg/l) still did not show any impacts of the WRL Site chloride plume in the shallow groundwater west of Killbuck Creek.

Round 4 groundwater results for chloride at the northwest quadrant of the WRL Site confirmed the general trends noted for Round 3 data, as discussed above (Figure 4-24 and 4-25). The WRL Site chloride plume is present only in the upper portion of the saturated zone in the vicinity of well nest B15/B15R/B15P, as indicated by the low chloride concentration of 10 mg/L for the deeper well B15P (Figure 4-29). The chloride concentration observed at well P1 increased from 65 mg/L in Round 3 to 80 mg/L in Round 4, but is still lower than the Round 2 result of 176 mg/L. The chloride concentration observed in the Round 4 sample from well P3R (77 mg/L) was similar to the Round 3 concentration (72 mg/L), but still higher than the Round 2 results (46 mg/L). The decreased chloride concentration observed at well P4R from Round 2 (188 mg/L) to Round 3 (25 mg/L) was confirmed in Round 4 (28 mg/L). The decreased chloride concentration observed at well G116A, which is located west of Killbuck Creek, from Round 2 (99 mg/L) to Round 3 (39 mg/L) was also confirmed in Round 4 (38 mg/L). The shallower well in well nest G116/G116A had still not shown any effects of the WRL Site chloride plume, as indicated by the Round 4 chloride result of 7 mg/L.

The second area exhibiting elevated chloride levels is in the vicinity of wells G110 and G114. The Round 1 and Round 2 chloride results for well G110 (166 mg/L and 234 mg/L, respectively) indicate the groundwater in the vicinity of G110 has been affected by WRL leachate. Round 1 and Round 2 samples from neighboring wells did not contain elevated chloride concentrations, indicating that the presence of leachate was quite limited (Figures 4-22, 4-23, 4-30, and 4-31). The chloride concentrations for nearby wells B13, P6, and G114 for Rounds 1 and 2 are 28 mg/L, 16 mg/L, and 42 mg/L; and 32 mg/L, 18 mg/L and 44 mg/L, respectively. The chloride anomaly at well G110 has been previously attributed to surficial leachate seeps along the southern slope. More recently, it was learned that the leachate-hauling trucks were loaded near well G110 at the base of the slope. Both of these conditions could have contributed to the presence of chlorides at well G110. The seeps are currently under control and leachate is now loaded on top of the landfill.

However, chloride concentrations have continually increased at well G110 from Round 1 to Round 4 (166 mg/L, 234 mg/L, 379 mg/L, and 523 mg/L, respectively), indicating increasing influence of WRL leachate on the groundwater in the vicinity over time (Figure 4-33). In addition, the chloride concentration at nearby well G114 increased from 44 mg/L in Round 2 to 176 mg/L and 134 mg/L in Rounds 3 and 4, respectively, clearly indicating the impacts of WRL leachate now include groundwater in the vicinity of well G114 as well as well G110 (Figures 4-22 to 4-25, 4-30 and 4-32). This influence of WRL leachate appears to be limited to areas close to the WRL Site margin upgradient of well nest nest B13/P6. Samples from well B13 (28 mg/L to 34 mg/L for Rounds 1 through 4), which is south of the WRL Site southern access road, show little change in chloride levels over Round 1 to 4 and are low in comparison to those of well G110. Well P6, the deeper well in nest B13/P6, also shows little change in chlorides from Rounds 1 to 4 (16 mg/L, 18 mg/L, 16 mg/L, and 20 mg/L, respectively) and are low in comparison to those of well G110 (Figure 4-32).

Chloride levels observed in samples from well G115 located at the southwestern margin of the WRL Site increased from 48 mg/L to 178 mg/L from Round 2 to Round 3, clearly indicating the presence of WRL leachate in Round 3. This trend persisted in the Round 4 chloride data for well G115 (191 mg/L)(Figures 4-22 and 4-25), indicating an increasing presence of WRL leachate at the southwestern margin.

The private water supply well (identified as PW1 in Round 1 and PWO in Round 2) results were 11 mg/l chlorides in Round 1 and 13 mg/l chlorides in Round 2. See Drawing 13160-F1 for the well location.

The pH measurements were generally lower where alkalinity is higher (Figures 4-36 and 4-37). The pH measurements varied over a narrow range. The pH of Round 1 samples ranged from 7.73 units (B9, B16A) to 6.41 units (P1). The pH of Round 2 samples ranged from 7.51 units (P6) to 6.51 units (G114). The pH of Round 3 samples ranged from 7.53 units (G111) to 6.53 units (G115). The pH of Round 4 samples ranged from 7.3 units (G116A) to 6.3 units (G115).

Alkalinity measurements typically are an indication of the amount of bicarbonate/carbonate equivalents present, but may also include other titratable species. Alkalinity measurements ranged from 245 mg/l (B16A) to 1270 mg/l (B15) in Round 1, from 238 mg/l (B16A) to 1640 mg/l (B15) in Round 2, from 246 mg/l (G111) to 1170 mg/l (B15R) in Round 3, and from 255 mg/l (G111) to 1060 mg/l (B15R) in Round 4. Wells with elevated levels of alkalinity, but not chlorides, include: B4, B10, B11, B11A, B12, B13, G108, G109, G109A, G112, G113, and G113A (Figures 4-22 and 4-34). Each of these wells, except B13, are hydraulically up- or sidegradient of the WRL Site. This indicates that WRL leachate is not responsible for the alkalinity anomalies where elevated chlorides are not present, which is consistent with the upgradient location of most of these wells.

Elevated chlorides, as discussed above, are a reliable indicator of the presence of WRL leachate. Some wells with elevated chlorides are associated with elevated alkalinity and depressed pH, but there are also wells with elevated alkalinity, depressed pH, and low levels of chlorides. This indicates that alkalinity and pH alone are not reliable in discriminating between WRL leachate affected vs unaffected groundwater, since elevated chlorides are not uniquely associated with these geochemical anomalies.

Specific conductance measurements for Round 1 samples ranged from 410 umhos/cm (G118A) to 4,390 umhos/cm (B15). Round 2 specific conductance measurements ranged from 550 umhos/cm (B6D) to 5,620 umhos/cm (B15). Specific conductance measurements for Round 3 samples ranged from 535 umhos/cm (P3R) to 3,840 umhos/cm (B15R). Round 4 specific conductance measurements ranged from 610 umhos/cm (B15P) to 3,590 umhos/cm (G110). Specific conductance of samples from wells exhibiting elevated chlorides were elevated. Elevated specific conductances were noted for WRL leachate affected groundwater (i.e., elevated chlorides) and groundwater with elevated alkalinity and low chlorides, so specific conductance measurements alone are of little use in discriminating between WRL leachate affected and unaffected groundwater (Figures 4-22 and 4-35).

Although not quantitatively determined, it is possible that slightly higher carbon dioxide concentrations due to landfill gas adjacent to the landfill could be a potential cause for the elevated alkalinity and specific conductance at the landfill borders. Because elevated alkalinity, specific conductance, and depressed pH are found at well B4, and elevated alkalinity has also recently been reported for upgradient well STI-6S (545 mg/L) (September 25, 1990 letter, A. Hiltner to G. Parker), it appears that upgradient conditions should also be evaluated further with regard to these geochemical anomalies.

Cyanide analysis was performed on samples collected from selected wells during Round 1 (see Section 2.1.2.1). No clear upgradient versus downgradient trend in cyanide concentrations was evident. The highest cyanide concentration was observed in a sample from well G116 (494 ug/L), a shallow well in the Killbuck Creek floodplain that does not exhibit elevated chlorides or alkalinity, and is thought to be unaffected by WRL leachate. Lower levels were observed in samples from the upgradient wells G109 (15 ug/L) and G109A (8 ug/L). Round 1 cyanide concentrations observed in wells affected by WRL leachate ranged from 6 ug/L (B15) to 193 ug/l (MW106). Cyanide was not detected in Rounds 3 or 4 samples from well G109. Cyanide was not detected in the Round 3 samples from well G109A, but was detected at 13 ug/L in Round 4. Cyanide concentrations for Rounds 3 and 4 samples from well G116 were 37 ug/L and not detected, respectively, indicating a decrease over the Round 1 samples of 494 ug/L. The Rounds 3 and 4 cyanide concentrations observed in wells affected by WRL leachate ranged up to 238 ug/L (B15R, Round 3). Since cyanide is present in the WRL leachate, the cyanide detected within the WRL leachate plume may be due to the WRL leachate.

Total phenolics were not detected in Round 1 samples. In Round 2, total phenolics were detected at only two locations, B15 (145 ug/L) and G110 (170 ug/L), both of which are associated with elevated chlorides, indicating WRL leachate is a potential source. Total phenolics were not detected in any Round 3 or 4 samples.

Calcium concentrations ranged from 225 mg/L (G109) to 46.2 mg/L (B15). Magnesium concentrations ranged from 107 mg/L (G109A) to 25.8 mg/L (G116). Sodium concentrations ranged from not detected (G115, G116) to 280 mg/L (B15). Potassium concentrations ranged from not detected (G109, G109A, G115, G116, G116A, G117, G118A, and B15P) to 141 mg/L (B15).

Iron and manganese concentrations tended to be higher in samples from wells thought to be affected by WRL leachate, but not without exception, indicating that these two reactive constituents are of limited value for discriminating between water types at the Site. Iron concentrations ranged from not detected (B15P, P4R, G116, G116A, G117, and G118A) to 11 mg/L (G115) (Iron was detected in the duplicate sample for well P4R), and manganese concentrations ranged from not detected (G116, G116A, and G118A) to 2.01 mg/L (G115).

Other metals analyzed for during Round 1 groundwater sampling are summarized below:

- Cobalt was detected at well P1 (84 ug/L) and in the duplicate sample at well P4R (63 ug/l).
- Copper was detected only once at well G116 at 122 ug/L.
- Lead was detected at only three wells, P4R (32 ug/L), G116A (6 ug/L), and G109 (7 ug/L).
- Nickel was detected at wells B15 (224 ug/L) and MW106 (213 ug/L), indicating nickel is present in groundwater unaffected by WRL leachate, but often found at higher levels in wells affected by WRL leachate.
- Silver was detected only at wells G109A (2 ug/L), P1 (3 ug/L), and P3R (3 ug/L).
- Thallium was detected in all samples ranging from 2 to 6 ug/L.
- Vanadium was only detected at wells G109 (50 ug/L) and G115 (60 ug/L).
- Zinc concentrations ranged from not detected (P1, G116A) to 6.34 mg/L (G109A).

Round 1 groundwater samples were also analyzed for aluminum, antimony, berylium, chromium (total), mercury, and selenium, none of which were detected. Round 3 groundwater samples were analyzed for total chromium and hexavalent chromium. Hexavalent chromium was not detected. Total chromium was detected in samples from B15P (1.4J ug/L), B15R (3.0 ug/L), G109A (3.5 ug/L), and G110 (2.4 ug/L).

Arsenic, barium, and cadmium were analyzed for in all four rounds of groundwater sampling from selected wells. Arsenic was detected in upgradient well B4 both times the well was sampled (10 ug/L, Round 1; 11 ug/L, Round 2), and found in a sample from well STI-5D at 13 ug/L (August 17, 1988; STI Report, 1990). Arsenic has also been detected in wells west of Lindenwood Road (B15, B15R, G109A, G110, G111, G114, G115, P1, and P3R), ranging up to 46 ug/L (B15, Round 1). It is possible that the arsenic detected in some groundwater wells west of Lindenwood Road are reflective of natural or anthropogenic (man-made influences) background. The elevated arsenic levels in wells adjacent to or east of Lindenwood Road could also be related to the depressed pH levels, where such a condition mobilizes naturally occurring arsenic in soil or rock. Arsenic levels in the WRL leachate (estimated values up to 318 ug/L) indicate that the WRL Site could also be a source of this constituent in groundwater downgradient. Arsenic was not detected in the furthest downgradient well nest G116/G116A. Arsenic was not detected in the samples from the private well (PW1, Round 1; PWO, Round 2). See Drawing 13160-F1 for location of the private well.

The results of barium analyses of groundwater samples follow a similar pattern as that for arsenic, where elevated levels are often, but not exclusively associated with samples exhibiting elevated chloride levels. For example, elevated barium levels were found in samples from wells B4 (Round 1, 1570 ug/L and Round 2, 1640 ug/L), G111 (Round 2, 218 ug/L), and G118R (Round 2, 467 ug/L), which were not associated with elevated chloride levels. Furthermore, the wells are located hydraulically up- or side-gradient of the WRL Site. The detection of barium in wells without elevated chlorides and/or upgradient of the WRL Site indicate the barium could be naturally occurring or related to an anthropogenic background. Low levels of barium were detected in the private well (PW1-Round 1, 49 ug/L; PWO-Round 2, 63 ug/L). Barium levels in the WRL leachate up to 4,710 ug/L indicate that the WRL Site could be a source of this constituent in groundwater downgradient of the Site.

Most results for cadmium were from not detected to 2.2 ug/L. Cadmium was found at upgradient as well as at downgradient locations. Samples with cadmium concentrations greater than 2.2 ug/L include the Round 1 sample from well G109 (16 ug/L), the Round

1 sample from well B15R (8 ug/L), and the Round 1 sample from well P3R (9 ug/L). The presence of cadmium was not confirmed in Rounds 2, 3 or 4 samples from wells G109 and B15R. Cadmium was detected in the Round 2 sample from well P3R (0.4 ug/L), but not confirmed in Rounds 3 or 4. Cadmium was not detected in the private well in Round 1 (PW1) or Round 2 (PWO).

The WRL leachate plume tends to contain elevated chlorides, sodium, potassium, magnesium, manganese, and iron. Other constituents sometimes associated with the WRL leachate plume include: total phenolics, cyanide, arsenic, barium, cobalt, copper, lead, nickel, silver, vanadium, and zinc. These latter constituents were either not detected or were present at only low levels in samples from well G116A, indicating that these constituents are attenuated in the aquifer.

4.4.2 Volatiles

VOCs detected in groundwater samples collected during the RI can generally be classified into the following groups: chlorinated ethenes (tetrachloroethene (PCE); trichloroethene (TCE); 1,1-dichloroethene; 1,2-dichloroethene (DCE); and vinyl chloride (VC)); chlorinated ethanes (1,1,2,2-tetrachloroethane, 1,1,1-trichloroethane, 1,2-dichloroethane, 1,1-dichloroethane, and chloroethane), 1,2-dichloropropane, and BETX (benzene, ethylbenzene, toluene, and xylenes). The dominant groups of VOCs tend to be the chlorinated ethenes, and to a lesser extent, chlorinated ethanes. Compounds from each of the major VOC groups described above were detected in samples from wells at the WRL Site and from upgradient wells. The greatest concentration of total VOCs in groundwater samples collected for the RI (Rounds 1 and 2) was at well B4, which is located upgradient of the WRL Site.

The detection of VOCs from the east to west direction (i.e., direction of groundwater flow) based upon the RI samples is as follows. The VOCs detected at well B4 (hydraulically upgradient of the WRL Site) include chlorinated ethenes, chlorinated ethanes, 1,2-dichloropropane, chloromethane, and BETX. The VOCs associated with wells upgradient of WRL near Lindenwood Road but not found at well B4 based upon RI samples include chlorobenzene, trans-1,3-dichloropropene, and dibromochloromethane. The VOCs detected only in locations downgradient of the WRL Site include low levels of carbon tetrachloride, bromoform, chloroform bromodichloromethane and acetone.

Four Rounds of groundwater samples were collected as described in Section 2. Rounds 1 and 2 included wells from east and west of Lindenwood Road. Groundwater samples were collected from selected wells west of Lindenwood Road in Rounds 3 and 4.

Chlorinated ethenes: Figures 4-38 and 4-39 are maps of the chlorinated ethenes (PCE, TCE, DCE, and VC) in Rounds 1 and 2 groundwater samples. (Note: The terms "total chlorinated ethenes, chlorinated ethenes, or ethenes" used in text or figures means the sum of the concentrations of PCE, TCE, DCE and VC, but exclusive of the ethene 1,1-dichloroethene.) Based upon Round 1 and 2 groundwater samples, the highest total chlorinated ethenes concentration was found at well B4 (2133 ug/L and 1912.48 ug/L, Round 1 and 2, respectively) with the second highest level found just west of Lindenwood Road south of the WRL Site at well G113A (309.39 ug/L and 532.6 ug/L, respectively). Both areas are upgradient of the WRL Site, and not associated with elevated levels of chlorides, indicating WRL leachate is not the source of these VOCs.

Total chlorinated ethenes were detected in Round 1 samples from the following wells east of Lindenwood road: B4 (2133 ug/L), B6D (6.6 ug/L), MW105 (1.09 ug/L), B7 (0.4 ug/L), B16 (29.9 ug/L), B16A (33.27 ug/L), B11 (1.65 ug/L), B11A (14.1 ug/L), G108 (27.65 ug/L, B10 (23.6 ug/L), and G112 (1.05 ug/L) (Figure 4-38). Total chlorinated ethenes were not detected in Round 1 samples from wells B6S and B9.

Total chlorinated ethenes were detected in Round 2 samples from the following wells east of Lindenwood Road: B4 (1912.48 ug/L), B6S (0.51 ug/L), B6D (88.58 ug/L), MW105 (0.37 ug/L), B16 (21.7 ug/L), B16A (38 ug/L), B11 (0.76 ug/L), B11A (16.4 ug/L), G108 (16 ug/L), B10 (27.7 ug/L), and G112 (1 ug/L) (Figure 4-39). Total chlorinated ethenes were not detected Round 2 samples from wells B7 and B9.

These Rounds 1 and 2 groundwater results document the presence of higher levels of total chlorinated ethenes in the deeper groundwater upgradient of the WRL Site east of Lindenwood Road. Of the well nests located east of Lindenwood Road (B6S/MW105/B6D, B16/B16A, B11/B11A), the deeper wells (B6D, B16A, and B11A) exhibited the greater concentration of total chlorinated ethenes (Figures 4-40 and 4-41).

In the upgradient area just west of Lindenwood Road and south of the WRL Site (G109/G109A, G113/G113A, B12, G111), based upon Rounds 1 and 2 groundwater samples, the highest chlorinated ethenes concentrations were found in samples from wells screened at depth in the aquifer. The total chlorinated ethenes Round 1 results for this group of wells are as follows: G109 (24.11 ug/L), G109A (98.95 ug/L), G113 (34.37 ug/L), G113A (309.39 ug/L) and B12 (137.41 ug/L)(Figure 4-38). The total chlorinated ethenes Round 2 results for this group of wells are: G109 (18.6 ug/L), G109A (115.1 ug/L), G113 (24.03 ug/L), G113A (532.6 ug/L), B12 (115.8 ug/L) and G111 (57.1 ug/L)(Figure 4-39).

The highest total chlorinated ethenes concentrations adjacent to the southeast portion of the WRL Site were in samples from well G113A (Rounds 1 and 2), which is the deeper well in the most distant well nest from the landfill in this area (Figures 4-38, 4-39, and 4-41). The deeper well (G109A) in well nest G109/G109A again exhibited the higher chlorinated ethenes concentrations (33 ug/L / 107 ug/L; and 4 ug/L / 73 ug/L, respectively) during Rounds 3 and 4. Chlorinated ethenes concentrations in Round 1 and Round 2 samples (Round 1; Round 2) from wells G112 (1.05 ug/L; 1 ug/L), G109 (24.11 ug/L; 18.6 ug/L) and B12 (137.41 ug/L; 115.8 ug/L) show a wide variation in concentrations for these shallow zone wells.

Round 2 results for a sample from well G111, which is a water table well about 650 ft southeast of the WRL Site, had a chlorinated ethenes concentration of 57.1 ug/L (Figures 4-39). Well G111 is located in a side-gradient position with respect to the WRL Site. Rounds 3 and 4 results from well G111 confirm the presence of chlorinated ethenes in the groundwater in that area (59 ug/L and 50 ug/L, respectively). The chloride concentrations observed at well G111 ranged from 22 mg/L to 26 mg/L between Round 2 to Round 4.

The chlorinated ethenes concentrations observed at well G114 decreased from Rounds 2 to 3 and 4 (14.75 ug/L, 11.55 ug/L, 3 ug/L and 1 ug/L Rounds 1 to 4, respectively). The observed chloride concentrations significantly increased from Rounds 2 to 3 and 4.

The chlorinated ethenes concentrations observed at well G110 from Rounds 1 to 4 (23.03 ug/L, 126.6 ug/L, 8 ug/L and 25.6 ug/L) show little relationship to the trend of increasing chloride concentration (Figure 4-33). The chlorinated ethenes concentrations observed at well nest B13/P6 have generally decreased from Round 1 to 4 (242.3 ug/L / 87.1 ug/L, 268.1 ug/L / 160 ug/L, 219 ug/L / 51 ug/L, and 215 ug/L / 78 ug/L, respectively).

The chlorinated ethenes concentrations observed at well G115 decreased from Rounds 2 to 3 and 4 (21.36 ug/L, 22.6 ug/L, 1 ug/L and 8 ug/L, Rounds 1 to 4 respectively). The observed chloride concentrations significantly increased from Round 2 to Rounds 3 and 4. This suggests that leachate affected water did not contain VOCs and was displacing the VOC contaminated groundwater at well G110, G114, and G115.

Total chlorinated ethenes were not detected in Round 1 or 2 samples from well B14.

Figure 4-41 is a plot of Round 2 total chlorinated ethenes along transect D-D', a cross-section along the southern boundary of the Site (see Figure 4-26 for cross-section location). Comparison with Figures 4-31 and 4-41 of chloride content reveals there is no apparent relationship between the chloride "leachate tracer" and the presence of total chlorinated ethenes. Chlorinated ethenes are present both inside and outside of the zone of elevated chlorides. These plots also show that the total chlorinated ethene concentrations vary considerably with location as shown by the total chlorinated ethene results of wells G113A (532.6 ug/L) and G109A (115.1 ug/L). This irregular distribution could be the representation of preferential flow in the fractured rock, as discussed in Section 4.3.

Total chlorinated ethenes were detected in Rounds 1 and 2 samples from well G107 located north of the WRL Site (2.21 ug/L and 10.15 ug/L, respectively)(Figures 4-38 and 4-39). The chloride concentrations were 20 mg/L and 16 mg/L, respectively (Figures 4-22 and 4-23).

The total chlorinated ethenes concentrations observed in samples from well B15R from Rounds 1 to 4 were 22.78 ug/L, 19.83 ug/L, 6.6 ug/L, and 10 ug/L and the associated chloride results were 477 mg/L, 348 mg/L, 529 mg/L, and 459 mg/L, respectively. The total chlorinated ethenes concentrations observed in the deeper well, B15P, in the well nest B15/B15R/B15P, from Rounds 1 to 4 were 17.28 ug/L, 15.2 ug/L, 15 ug/L, and 18 ug/L, respectively, with associated chloride concentrations of 14 mg/L, 15 mg/L, 13 mg/L, and 10 mg/L. The chloride concentrations indicate the WRL leachate plume is present in the vicinity of well B15R, but not B15P.

The total chlorinated ethenes concentrations observed at well nest P3R/P4R indicated the highest total chlorinated ethenes concentrations were consistently associated with the deeper well, P4R. Results from Rounds 1 to 4 for well nest P3R/P4R were (16.24 ug/L / 53.60 ug/L, 12.14 ug/L / 47.07 ug/L, 6 ug/L / 30 ug/L, 7 ug/L / 30 ug/L, respectively). This same trend was not noted in the chloride concentrations observed at well nest P3R/P4R, since in Rounds 1 and 2 samples from well P4R had the highest chloride concentrations, and in Rounds 3 and 4, samples from well P3R had the highest chloride concentrations (47 mg/L / 149 mg/L, 46 mg/L / 188 mg/L, 72 mg/L / 25 mg/L, 77 mg/L / 28 mg/L, respectively).

The chlorinated ethenes concentrations observed at well nest P1/MW106 for Rounds 1 and 2 showed little difference to slightly higher levels in the samples from the deeper well MW106 (Figure 4-38 and 4-39). The total chlorinated ethenes concentrations observed in samples from well P1 generally decreased from Round 2 to Rounds 3 and 4 (21.56 ug/L, 7 ug/L, and 9 ug/L, respectively).

Total chlorinated ethenes were not detected in samples from well G117, located northwest of the WRL Site, in Rounds 1 or 2 (Figures 4-38 and 4-39). Total chlorinated ethenes were also not detected in samples from wells G118R (Round 2), G118A (Round 2), or G119 (Rounds 1 and 2). Total chlorinated ethenes were detected in the deeper wells in the well nests G119/G119A and G118R/G118A (G119A-Rounds 1 and 2; G118A-Round 1, only) (Figures 4-38 and 4-39).

Total chlorinated ethenes were not detected in samples from well G116 in any of the four rounds. Well G116 is the shallow well in the well nest G116/G116A located west of Killbuck Creek. The chlorinated ethenes were detected in the deeper well, G116A, in each of the four rounds (12.34 ug/L, 27.5 ug/L, 12 ug/L, and 22 ug/L).

Along transect C-C′, a cross-section to the north of the WRL Site (see Figure 4-26 for location), Round 2 chlorinated ethenes are present upgradient of the WRL Site and extend below and past the WRL Site (Figure 4-40). Comparison of these plots with chloride results depicted in Figures 4-23 and 4-28 reveal the presence of chlorinated ethenes both inside and outside of the WRL leachate chloride plume. Warzyn believes this indicates that the WRL leachate plume is overprinting a chlorinated ethene plume.

VOCs are present hydraulically upgradient of the WRL Site and are affecting the water quality beneath the landfill. Figure 4-42 depicts a total VOC plume along transect C-C′ being present throughout the aquifer area studied adjacent to the WRL Site, and overprinted by a WRL leachate plume originating from beneath the landfill based upon Round 2 results. (Note: Total VOCs is the sum of all VOCs detected.) Comparison of Figures 4-40 and 4-42 shows that total VOC pattern in the groundwater is consistent with that of total chlorinated ethenes results.

Simple dilution calculations for chlorinated ethenes in leachate versus groundwater (Rounds 1 and 2 data) were performed. A similar calculation was made for the chloride data (Table 4-5). Chloride appears more highly diluted in the groundwater. This is not consistent with the expected relative behavior of reactive chlorinated ethenes versus conservative chlorides (i.e., chlorinated ethenes should exhibit greater attenuation than chlorides), further suggesting that WRL leachate is not the source of chlorinated ethenes in groundwater where chlorides are observed at low levels. Based on the chloride dilutions noted, observed levels of chlorinated ethenes present in WRL leachate should be reduced to nearly non-detectable levels during transport to the furthest downgradient wells.

Chlorinated ethanes: Based upon Rounds 1 and 2 data, 1,1-dichloroethane, 1,1,1-trichloroethane, chloroethane, and 1,2-dichloroethane are the most widely detected chlorinated ethanes. The chlorinated ethanes generally follow the distribution pattern of the chlorinated ethenes described above, but at lower concentrations. The largest total chlorinated ethane concentrations (1,1-dichloroethane, 1,1,1-trichloroethane, chloroethane, 1,2-dichloroethane, 1,1,1,2-tetrachloroethane, and 1,1,2-trichloroethane) were observed at the upgradient well B4 in both Rounds 1 and 2 (503.98 ug/L and 541.3 ug/L, respectively). The "southeast corner area" at the WRL Site exhibited the second highest levels of total chlorinated ethanes. Chlorinated ethanes were detected in the groundwater west of Killbuck Creek, but only in the deep well G116A in the well nest G116/G116A.

The compound 1,1-dichloroethane was detected in Rounds 1 or 2 groundwater samples from 27 wells: B4, B6D, B10, B11A, B12, B13, B15, B15P, B15R, B16, B16A, G107, G108, G109, G109A, G110, G111, G113, G113A, G114, G115, G116A, MW106, P1, P3R, P4R, and P6. 1,1-Dichloroethane was again detected in Rounds 3 or 4 samples from wells B13, B15P, B15R, G109, G109A, G110, G111, G114, G115, G116A, P1, P3R, P4R, and P6. 1,1-Dichloroethane was not detected in the samples from well G116 in Round 3 or 4 . The other wells listed above were not sampled in Rounds 3 or 4.

The compound 1,1,1-trichloroethane was detected in Rounds 1 or 2 groundwater samples from 26 wells: B4, B6S, B6D, MW105, B7, B10, B11, B11A, B13, B15P, B15R, B16, B16A, G107, G108, G109A, G110, G111, G113A, G116A, G119A, MW106, P3R, P4R, P6, and P7. 1,1,1-Trichloroethane was again detected in Rounds 3 or 4 samples from wells B13, B15P, G111, G116A, P4R, and P6. 1,1,1-Trichloroethane was not detected in Rounds 3 or 4 samples from B15R, G109, G109A, G110, G114, G115, G116, P1, and P3R. The other wells listed above for Rounds 1 and 2 were not sampled in Rounds 3 or 4.

Chloroethane was detected in Rounds 1 or 2 groundwater samples from 20 wells: B4, B10, B11A, B12, B13, B15, B15R, G108, G109, G109A, G110, G113, G113A, G114, G115, G116A, MW106, P1, P3R, and P4R. Chloroethane was again detected in Rounds 3 or 4 samples from wells B15R, G110, G115, P1, and P3R. Chloroethane was not detected in Rounds 3 or 4 samples from wells B13, B15P, G109, G109A, G111, G114, G116, G116A, P4R, and P6. The other wells listed above were not sampled in Rounds 3 or 4.

The compound 1,2-dichloroethane was detected in Rounds 1 or 2 groundwater samples from 15 wells: B4, B6D, B12, B13, B15, B15R, G109A, G110, G113A, G114, MW106, P1, P3R, P4R, and P6. 1,2-Dichloroethane was again detected in Rounds 3 or 4 samples from wells B13, B15R, G109A, G110, and G114. 1,2-Dichloroethane was not detected in Rounds 3 or 4 samples from wells B15, G109, G111, G115, G116, G116A, P1, P3R, P4R, and P6. The other wells listed above were not sampled in Rounds 3 or 4.

Chlorinated ethanes were not detected in Round 1 or 2 samples from wells as follows: B6D (Round 1), B7 (Round 2), B9 (Rounds 1 and 2), B11 (Round 2), B14 (Rounds 1 and 2), G112 (Round 2), G116 (Rounds 1 and 2), G117 (Rounds 1 and 2), G118R (Round 1 not tested, Round 2), G118A (Rounds 1 and 2), and G119 (Rounds 1 and 2).

Benzene, ethylbenzene, toluene, xylene (BETX): Based upon Rounds 1 and 2 groundwater samples, benzene, ethylbenzene, toluene, and total xylenes were detected both upgradient and downgradient of the WRL Site. The highest total BETX concentration observed in Round 1 was in the groundwater sample from well B4 (48.3J ug/L), but the highest BETX concentration found in Round 2 was in the groundwater sample from well G110 (31.8J ug/L) ("J" indicates concentration is estimated).

Benzene was detected in Rounds 1 or 2 groundwater samples from 15 wells: B4, B12, B13, B15, B15R, G109A, G110, G113, G113A, G114, G115, MW106, P1, P3R, and P4R. Of the wells sampled in Rounds 3 and 4, benzene was consistently confirmed, except for wells P1 and P3R, where benzene was not detected in Round 4. In addition, benzene was detected once during the four rounds of sampling in the Round 3 sample from well G109 (0.5J ug/L).

Ethylbenzene was detected in Rounds 1 or 2 groundwater samples from 14 wells: B4, B12, B13, B15, G109, G109A, G110, G113, G113A, G114, G115, MW106, P3R, and P4R. Of the wells which were sampled in Rounds 3 and 4, ethylbenzene was only detected in the sample from wells G110 (9 ug/L) and B15R (0.8J ug/L) and ethylbenzene was not detected in samples from wells B13, G109, G109A, G114, G115, P3R, and P4R.

Based upon Round 1 groundwater samples, chloromethane was detected in only three wells (B4, P3R, and P4R). These detections were not confirmed in Round 2, or Rounds 3 and 4 (for P3R and P4R).

Carbon tetrachloride was detected only once in Round 1 at well P6 (0.23 ug/L) and 6 times in Round 2 (P6, B13, B15P, G114, G115, and G117). Carbon tetrachloride was not detected in either the Rounds 3 or 4 groundwater samples from wells P6, B13, B15P, G114, and G115 (G117 was not sampled in Rounds 3 or 4).

Bromoform was detected only once at well P1 (0.49 ug/L; Round 1).

Trans-1,3-dichloropropene was detected in five wells in Round 1 (B10, B12, B13, B16, and G113A) and detected in only two wells in Round 2 (G113A and G114). Of the wells sampled in Rounds 3 and 4, trans-1,3-dichloropropene was not detected.

Bromodichloromethane was detected only in the Round 1 duplicate sample from well PW1 (0.21J ug/L) and in the Round 2 sample from well B15 (0.24J ug/L). Of the wells sampled in Rounds 3 and 4, bromodichloromethane was not detected. Dibromochloromethane was detected only twice in Round 1 groundwater samples (G113A and G114) and not detected in Rounds 2, 3, or 4.

Methylene chloride (dichloromethane) was not detected in Round 1 or Round 2 samples. Of the wells sampled in Round 3, methylene chloride was not detected. Methylene chloride was detected in samples from three of the wells sampled in Round 4, G109A (1J ug/L), G110 (3J ug/L), and B15R (3J ug/L), all at estimated concentrations (J). Acetone was not detected in Round 1 or 2 GC/MS confirmation samples (acetone is not a Hazelton GC target compound). Acetone was not detected in the Round 3 groundwater samples. Acetone was detected three times in Round 4: B13 (6J ug/L) B15R (11 ug/L), and G114 (8J ug/L).

Private Well: One private water supply well, identified as PW1 in Round 1 and PWO in Round 2, located southwest of the WRL Site was sampled in Rounds 1 and 2. No VOCs were detected in samples from this well in either round. Bromodichloromethane was detected in the Round 1 duplicate sample from well PW1 at 0.21J ug/L. See Drawing 13160-F1 for the location of the well (PW1).

4.4.3 Semi-Volatiles

Based upon Round 1 groundwater samples, the following semi-volatile compounds (SVOCs) were detected: acenaphthene (B15; 0.6J ug/L), dibenzofuran (B15; 0.3J ug/L); bis (2-ethylhexyl) phthalate (MW106; 7J ug/L) (P1; 5J ug/L), 1,2-dichlorobenzene (B15; 3J ug/L), and 1,4-dichlorobenzene (B15; 11 ug/L) (B15R; 6J ug/L) (G109; 9J ug/L) (G109A; 3J ug/L) (G115; 4J ug/L) (MW106; 12 ug/L) (P1; 5J ug/L).

The compound list for the Rounds 1 and 2 groundwater volatile compound analytical method overlaps the compound list for the semi-volatile analytical method for the compounds 1,2-dichlorobenzene, 1,3-dichlorobenzene and 1,4-dichlorobenzene, so these compounds are addressed here as semi-volatile compounds. Based upon Round 1 and 2 volatiles method data, 1,2-dichlorobenzene was detected in 11 wells: B12, B13, B15, B15R, G109, G109A, G110, G113A, G114, MW106, and P1. 1,2-Dichlorobenzene was not detected in Round 3 (semi-volatile method) groundwater samples from wells: B15P, B15R, G109, G109A, G110, P1, or B13 (B12, MW106, and G113A were not sampled in Round 3), but was detected in the sample from well G114 (4J ug/L). 1,3-Dichlorobenzene was detected only once in the Round 2 groundwater sample from well G110 (0.44 ug/L).

Based upon Rounds 1 and 2 volatiles method data, 1,4-dichlorobenzene was detected in 17 wells: B10, B12, B13, B15, B15R, G108, G109, G109A, G110, G113, G113A, G114, G115, MW106, P1, P3R, and P4R. 1,4-Dichlorobenzene was not detected in Round 3 (semi-volatile method) groundwater samples from wells G109, P1, P3R, and P4R (B10, B12, B15, G108, G113, G113A, and MW106 were not sampled in Round 3), but was observed in samples from wells B13, B15R, G109A, G110, G114, and G115.

No SVOCs were detected in RI samples from well B4. The general group of SVOCs associated with wells upgradient of the WRL near Lindenwood Road include: 1,2-dichlorobenzene, 1,4-dichlorobenzene and bis (2-ethylhexl) phthalate. The SVOCs detected only in sample locations downgradient of the WRL Site include: acenaphthene, dibenzofuran, and 1,3 dichlorobenzene. No pesticides or PCBs were detected in any of the RI groundwater samples.

Tentatively identified compounds were detected in samples from wells B13, B15, B15R, G109A, G110, G114, G115, MW106, P3R, and P4R, at estimated concentrations less than 50 ug/L, with the exception of sulfur, which ranged up to 650 ug/L (G110).

4.4.4 Pesticides/PCBs

Results of pesticide and PCB analyses (Round 1) showed no target compounds at detectable levels.

4.4.5 Summary

There are three general areas where chloride concentrations are elevated indicating the presence of WRL Site leachate. These are:

- The northwest quadrant of the WRL Site defined by wells B15R, MW106, P1, P4R, and G116A
- In the vicinity of wells G110 and G114
- In the vicinity of well G115

The RI results of the private well sampling indicate it is not being impacted by inorganic constituents or VOCs.

The WRL leachate plume tends to contain elevated levels of chlorides, sodium, potassium, magnesium, manganese, and iron. Other constituents sometimes associated with the WRL plume include total phenolics, cyanide, arsenic, barium, cobalt, copper, lead, nickel, silver, vanadium, and zinc. These constituents were either not detected or present at only low levels in samples from well G116A, indicating that these constituents may be attenuated in the aquifer, as expected.

The groundwater upgradient of the WRL Site has been impacted by VOCs with the highest concentrations being observed in samples from well B4. The RI report concludes that the WRL leachate plume has overprinted VOC impacted groundwater. The VOCs observed in the RI samples can be divided up into three general groups based upon the detection of VOCs in groundwater from the east to west direction (i.e., direction of groundwater flow);

- VOCs associated with well B4
- VOCs associated with wells upgradient of WRL near Lindenwood Road, and

- VOCs present only in the WRL leachate plume

The VOCs detected in RI samples at well B4 (hydraulically upgradient of the WRL Site) include chlorinated ethenes, chlorinated ethanes, 1,2-dichloropropane, chloromethane, and BETX (benzene, ethylbenzene, toluene, and xylenes). The VOCs associated with wells upgradient of the WRL Site near Lindenwood Road but not found at well B4 based upon RI samples include chlorobenzene, trans-1,3-dichloropropene, and dibromochloromethane. The VOCs detected only in locations downgradient of the WRL Site include low levels of carbon tetrachloride, bromoform, bromodichloromethane, and acetone.

No SVOCs were detected in RI samples from well B4. The general group of SVOCs associated with wells upgradient of the WRL near Lindenwood Road include: 1,2-dichlorobenzene, 1,4-dichlorobenzene and bis (2-ethylhexl) phthalate. The SVOCs detected only in sample locations downgradient of the WRL Site include acenaphthene, dibenzofuran, and 1,3-dichlorobenzene. No pesticides or PCBs were detected in any of the RI groundwater samples.

VOCs are present both inside and outside of the limits of the WRL leachate groundwater plume as defined by elevated chlorides. This indicates that the presence of VOCs outside of the WRL leachate plume are not due to WRL leachate. Potential other sources of VOCs to the groundwater include releases from the upgradient Acme Solvent Site, migration of landfill gas, and a localized source such as effluent from the household septic systems in the immediate vicinity. (See Section 4.7 for discussion.)

4.5 Surface Water and Sediment Analyses

Surface water (SW) and sediment (SD) samples were collected at five locations along Killbuck Creek (see Drawing 13160-F1). There were two rounds of surface water sampling and one round of sediment sampling. Four of the sampling locations are downstream of the landfill (SW01 through SW04 and SD01 through SD04). The furthest upstream location is considered to represent background conditions (SW05 and SD05).

4.5.1 Inorganics - Indicators

Surface Water: Surface water samples were analyzed for field pH, field specific conductance, total phenolics, chloride, alkalinity, arsenic, barium, and cadmium. Results for pH showed no upstream-downstream trends, but rather showed little variation, ranging from 7.81 to 7.89 units for Round 1, and 7.94 to 8.18 units for Round 2. Similarly, for specific conductance results, no upstream-downstream trends were noted as the results were relatively constant, ranging from 570 umhos/cm to 620 umhos/cm for Round 1 and 685 for all Round 2 samples. Alkalinity results ranged from 186 mg/L to 217 mg/L for Round 1 and 227 mg/L to 238 mg/L for Round 2. Chloride concentrations ranged from 25 mg/L to 27 mg/L for Round 1 and 42 mg/L to 43 mg/L in Round 2. Again, no upstream-downstream trends were noted in either the alkalinity or chloride results.

Arsenic and cadmium were not detected in the surface water samples. Arsenic was detected in the Round 2 duplicate sample SW05, the upstream location, at 3.40K ug/L. Barium was detected in all samples at concentrations ranging from 70 ug/L to 80 ug/L for Round 1 and 74.2K ug/L to 100K ug/L for Round 2. No upstream-downstream trends were noted in the Round 1 barium results. Total phenolics were detected only in the samples taken at SW05 (upstream background sample) at 7 ug/L in Round 1. Total phenolics were not detected in Round 2 samples. Total and hexavalent chromium were analyzed for in Round 2 and were not detected. Cyanide was detected in one samples (SW04) at 17J ug/L in Round 2. Cyanide was not analyzed for in Round 1.

The USGS maintains a surface water monitoring station on Killbuck Creek downstream from the WRL Site at the bridge on State Highway 251 (Figure 1-1). For the period from October 28, 1987 to September 15, 1988, the USGS collected nine samples and recorded the instantaneous discharge of the stream. The samples were tested by the USGS for a variety of parameters, including pH, specific conductance, hardness, calcium, magnesium, sodium, barium, and manganese (Appendix G). The parameter concentrations showed little correlation to the discharge rate of the stream. The pH, specific conductance, and barium concentrations were similar in value to the results of the samples of this study.

Since upstream-downstream trends in the results are not evident in the surface water samples from this study based upon the inorganic parameters, the WRL Site does not appear to be impacting the water quality of Killbuck Creek. In addition, comparison of the data from this investigation to the USGS data indicates that this conclusion is valid at other stream flow discharges, since the parameter concentrations (USGS data) do not vary with discharge. The WRL "leachate tracer", chloride, showed almost no change in concentration, indicating WRL leachate is not impacting the surface water.

Sediments: Sediment samples were analyzed for 23 metals and cyanide. The five samples locations are shown on Drawing 13160-F1. The results do not exhibit any upstream-downstream trends and downstream sample concentrations are generally comparable to those of the background sample (SD5). The sediment results for metals are also at or below the mean values for soils reported by Connor, Shacklette, et al. (1975), except for calcium and magnesium as would be expected due to the presence of dolomite bedrock at this Site. Pierce, et al. (1982) found total cadmium levels in Minnesota glacial drift to range from 0.06 ppm to 0.74 ppm, with the highest Cd concentration in calcareous soils, reflecting the ability of cadmium to substitute for calcium in the calcite crystal. Connor, Shacklette, et al (1975) conducted a study to determine the "large scale" or "ordinary" natural geochemical variation in rocks, soils, and plants in the conterminous United States. Since upstream-downstream trends are not evident and sediment concentrations of downstream samples are comparable to both the background sample (SD5) and reported values for dolomite and limestone (arsenic (<1 ppm - 39 ppm), barium (1 ppm - 3,000 ppm), cadmium (<1 ppm - 12 ppm), and lead (4 ppm - 18 ppm) (Connor, Shacklette, et al. 1975; Pierce, et al., 1982)), the WRL Site has not created inorganic impacts to the sediments in Killbuck Creek.

4.5.2 Volatiles

Surface Water: Results of Round 1 analyses indicate that VOCs are not likely present in stream waters at measurable levels. Results for surface water at location SW1 showed methylene chloride at 19 ug/L. Methylene chloride is a typical field/laboratory contaminant, which is likely to account for this result. VOCs were not detected in sample SW02, but the SW02 GC/MS confirmation sample contained 0.29 ug/L of chloroform. The only other surface water sample having measurable VOCs was collected at location SW03, where four VOCs at low concentration (within 1 to 2 ug/L of their detection limits) were detected. However, a duplicate sample collected at SW03 had no detectable VOCs. The field blank (FB04) shows two VOCs at low concentrations. Because the detection of VOCs was not duplicated and VOCs were present in the field blanks, these results probably reflect field/laboratory contamination. No VOCs were detected in Round 2 samples. Based on Round 1 and Round 2 VOC results, VOCs are not impacting the surface water. This is consistent with the chloride data which show no impact by the WRL leachate.

Sediments: Results of VOC analyses for sediment samples (Round 1) indicated chloroform may be present in two of the five samples (SD02 and SD04) at low concentrations (2 ug/kg). However, results are below method detection limits (i.e., the level of lowest numerical quantification) and are considered estimated. No other VOCs were detected.

4.5.3 Semi-Volatiles

Semi-volatiles (SVOCs) analyses were performed on sediment samples (Round 1). Except for sample SD05, only phthalates (bis(2-ethylhexyl)phthalate and di-n-butylphthalate) were detected. Phthalates ranged from not detected (SD04) to 4400 ug/kg (SD02). Phthalates tend to be ubiquitous due to their widespread use as plasticizers, and are reported to be naturally occurring in organic matter by Dragun (1988) (p. 343) ranging from 150 to 925 mg/kg for bis (2-ethyl hexyl) phthalate and 19 to 56 mg/kg for dibutylphthalate. At the upstream location SD05, several polycyclic aromatic hydrocarbons (PAHs) were detected at low concentrations (less than method detection limits).

4.5.4 Pesticides/PCBs

No pesticides or PCBs were detected in sediments (Round 1).

4.6 Ambient Air Quality

The ambient air sampling program consisted of one upwind location (Upwind #4) and three downwind locations (Downwind #1, #2, and #3) (Figure 2-2). The Upwind #4 was located northwest of the landfill and Downwind #1, #2 and #3 were located on the east half of the landfill towards the south side. The samples from Downwind #1 were damaged and not analyzed. The meteorological station was located in the approximate center of the landfill.

The winds were consistently out of the northwest with periods of calm conditions. Wind speeds ranged from calm to over 10 miles per hour. Air temperatures ranged from just over 3°C to more than 12°C.

The compounds detected at Upwind #4 were carbon tetrachloride, chloroform, ethylbenzene, 1,4-dichlorobenzene, methylene chloride, styrene, tetrachloroethene, toluene, trichloroethene, o-xylene and isopropylbenzene (Appendix C).

Each of the above listed compounds detected at Upwind #4 plus hexane, 1,2-dichloropropane, and m,p-xylenes were detected at Downwind #2. Hexane was detected in only two of the four samples collected at Downwind #2. 1,2-Dichloropropane was detected in only one of the four samples collected at Downwind #2 (Appendix C). Hexane, 1,2-dichloropropane, m,p-xylenes, and isopropylbenzene were not detected at Downwind #3. The four samples at Downwind #2 were collected at four different pumping rates.

Although a quantitative evaluation cannot be made due to the qualified nature of the data, (Appendix H) the data do not indicate the WRL Site is having a significant impact on the ambient air quality, since nearly the same group of compounds were detected at the upwind location as the downwind location. The range of concentrations detected in both up and downwind samples is listed in Table 4-7.

4.7 Potential Contributing Sources

VOCs are present in the groundwater both east and west of Lindenwood Road, but as discussed earlier, the WRL leachate plume, as characterized by its elevated chloride content and the major cations, sodium plus potassium, is limited to the northwest quadrant of the WRL Site, in the vicinity of wells G110 and G114, and in the vicinity of well G115. This indicates that the presence of VOCs outside of the defined WRL leachate plume are not due to WRL leachate. Potential other sources of VOCs to the groundwater include releases from the upgradient Acme Solvent Site, migration of landfill gas, and localized sources such as effluent from the household septic systems in the immediate vicinity.

4.7.1 Groundwater Releases from the Acme Solvent Site. The Acme Solvent Site was used for the disposal of solvent still-bottom sludges, non-recoverable solvents, paints and oils into unlined lagoons. It was also reported that an unknown number of drums were crushed and buried on the Acme Solvent Site (Ecology and Environment, 1983; Jordan, 1984). These disposal practices have resulted in the release of VOCs to the groundwater as documented by the detection of VOCs in groundwater samples from upgradient wells. The highest concentration of VOCs observed in the RI was in the groundwater samples from well B4 at the Acme Solvent Site, which is consistent with the disposal of solvent waste at the Acme Solvent Site. The Acme Solvent Site is located hydraulically upgradient from the WRL Site, and so releases to the groundwater at the Acme Solvent Site are expected to be transported in a downgradient direction towards the WRL Site.

The highest total VOC concentrations observed in RI groundwater sample results was from well B4, which is located upgradient of the WRL Site. The second highest level of VOCs was observed in samples from well G113A which is located just west of Lindenwood Road south of the WRL in an upgradient location. Between these two well locations (B4 and G113A), there is an area without observed high levels of VOCs. This apparent distribution of VOCs in the groundwater upgradient of the WRL Site may be an artifact of monitoring well placement, both horizontally and vertically, which means well screens may not be adequately located to intercept the VOC plume in this fractured dolomite aquifer.

In Section 4.3.4, the potential for preferential flow through the dolomite aquifer is discussed. Based upon geological data, geophysical data, pumping test results, permeability test results, water level observations, and groundwater quality data, it is believed that a high permeability zone exists in the vicinity of the elevation interval 690 ft to 710 ft MSL. Currently, none of the wells in the central area described above are monitoring the vicinity of the high permeability zone. For example, Figure 4-19 illustrates the approximate location monitoring well screens with respect to the vicinity of the high permeability zone. Also shown on Figure 4-19 are the groundwater flow paths showing how the existing water table wells and deep wells could miss the VOCs being transported between the Sites. The U.S. EPA stated the need to install more wells to further characterize the groundwater upgradient of the WRL Site in this central area (October 20, 1989 letter, Mr. B. Schorle to Mr. G. Marzorati), which indicates that the distribution of VOCs in this area is not adequately defined. It is recommended that an investigation be conducted in the central area to provide conclusive documentation of the presence or absence of VOCs in possible preferential pathways.

The Acme Solvent PRP group has installed two new wells (STI-5S and STI-7S) in this central area ("Northwest Area Investigation, Acme Solvent Reclaiming, Inc., Oct. 1990 (NWA, 1990)). The sample from well STI-7I (nested with well B9) contained 14.9 ug/L total VOCs, but no VOCs were detected in the sample from STI-5S. The result from STI-7I documents the presence of VOCs in the groundwater between the two sites (NWA, 1990) and is inconsistent with the conclusion that VOCs are "virtually absent" from this area. (See STI Report, p. ES-2 for characterization of "virtually absent".) With regards to the Northwest Area Investigation, the USGS recommended among other things, continuous coring of the dolomite aquifer to locate fracture zones, and sampling of discrete fractured intervals for VOCs to select the most appropriate location for the installation of a monitoring well (R. Kay letter to A. Hiltner, Dec. 30, 1989) in order to determine if the Acme Solvent Site is the sole source of VOCs in the area. However, it appears that this was not done. The well construction logs for well STI-5S and STI-7I (NWA, 1990) indicate the well intakes (i.e., open interval or screened interval) were not located in the fracture zone shown on Figure 4-19.

It has been suggested in the STI Report (1990) that a low permeability zone is present in the same area believed by Warzyn to contain the high permeability zone. This interpretation is not consistent with the potentiometric maps prepared for this report or the STI Report (1990). These potentiometric maps indicate the presence of a higher permeability area rather than a low permeability area. This comparison is illustrated in the hypothetical flow model shown in Figures 4-17 and 4-18. The configuration of potentiometric contours on maps prepared for this report and the STI Report (1990) appear to be consistent with the high permeability zone model (Figure 4-17).

The low permeability conclusion in the STI Report (1990) is based upon permeability testing from wells B6S, B6D, MW105, MW201B, STI-5I, and STI-5D, which all are deep wells except for B6S. B6S has the highest permeability of the above listed group of wells. Permeability results of the above listed wells were compared to the well depth to the permeability test results (STI Report, 1990) for wells G101, P8, P9, MW202, B4, B7, B16, B16A, B9, MW104, and STI-2I. Of these, six of the 11 wells are shallow wells and have higher permeability than the deeper wells (Table J-3, App. J, STI Report 1990). These data indicate that the shallow portions of the dolomite aquifer are more permeable than the deeper portions, which is consistent with the observation of spacing between bedding fractures increases with depth (p. 30, Volume 1, STI Report, 1990).

Additional support for a continuous, but channeled VOC plume between the two Sites is the distribution relationships between parent chlorinated ethenes and degradation products in the groundwater, especially the presence of the cis- isomer of 1,2-dichloroethene. The differentiation of isomers is important to the interpretation of the VOC transport in groundwater, since the presence of cis-isomer of 1,2-DCE is indicative of biologically mediated degradation of more chlorinated parent species. EPA methods typically used to analyze groundwater samples for VOCs do not differentiate between the cis- and trans-isomers of 1,2-dichloroethene (1,2-DCE) and the results are reported as the trans-isomer (Cline and Viste, 1985). Groundwater testing in the RI using a methodology that differentiates between the cis- and trans-1,2-DCE isomers has shown cis-1,2-DCE to be the dominant isomer present in the area groundwater. Wood, et al. (1981) determined that biodegradation of trichloroethene produces dominantly the cis-isomer of 1,2-DCE with traces of the trans-isomer, as well.

Vinyl chloride (VC), 1,1-dichloroethene, cis- and trans-1,2-dichloroethene, 1,1-dichloroethane, and chloroethane are either not commercially produced or are not in wide use (i.e., used in specialized industries). Parent compounds such as tetrachloroethene (PCE), trichloroethene (TCE), 1,1,1-trichloroethane, and methylene chloride (Wood, et al., 1981), are much more widely used, which further suggests that the presence of less chlorinated compounds in groundwater may be due to a degradation process. Biodegradation of chlorinated ethenes and ethanes in groundwater occurs under anaerobic conditions through the loss of a chlorine atom (Vogel, et al., 1987). Wood, et al., (1981) and Vogel and McCarty (1985) found PCE to degrade to TCE, then to DCE and finally to VC. Studies by Wilson, et al., (1986) and Barrio-Lage, et al., (1986) additionally determined that the cis-isomer of DCE degraded to chloroethane as well as VC.

The percent of an individual ethene to the total concentration of ethenes provides a gauge as to the relative level of degradation present. Dolomite wells east of Lindenwood Road generally have the highest percentage of PCE, while unconsolidated material wells and dolomite wells west of Lindenwood Road tend to have the highest percentages of VC, with no clear distinctions for TCE and DCE (Table 4-6). In addition, nearly all the DCE detected in Round 1 and Round 2 samples was the cis-isomer. The decrease in PCE as a percentage of total ethenes, the increase in VC as a percentage of total ethenes from east to west (downgradient), and the dominance of the cis-isomer of DCE, suggest that biodegradation of chlorinated ethenes may be occurring and playing a role in the distribution of VOCs in the groundwater.

Additional support indicating that the 1,2-dichloroethene in the southeast corner is related to downgradient transport and degradation is the correlation between 1,2-dichloroethene and total VOCs. Hickok (1985) reported that the correlation coefficient (r) between total VOC content and DCE content is +0.999 for the groundwater data from the Ecology and Environment (1983) report and the Jordan (1984) report. Both reports include wells east and west of Lindenwood Road. This indicated that DCE content is highly related to the VOC content of groundwater, rather than due to a second source as speculated by Jordan (1984). One would not expect such a high correlation between DCE and total VOC content, if there was a completely separate source of DCE to the groundwater. Hickok (1985) stated in their report that this implied ". . . in the general vicinity of the Acme site, groundwater contamination with VOCs is virtually co-definitional with groundwater contamination with DCE" (p. 1.18). Hickok (1985) stated that this relationship is contradicted by the sparse detection of DCE in test pit samples from the Acme Solvent site reported in the Jordan (1984) study, as follows:

Acme Solvents Site

Test Pit Samples	DCE (ug/kg)	Total VOC Content (ug/kg)
1 S-4	2,500	143,800
4 S-3	3,200	1,140,800
4 S-5	20,000	1,412,000
5 S-3	69	3,583

However, the DCE detected in groundwater samples may be a result of biodegradation, either in source area soil in the vadose zone prior to reaching the groundwater or in the groundwater, so it is perhaps more important to note that the assumed parent compounds (PCE, TCE) of DCE were found in measurable concentrations in all but three of the 15 test pits (pits 11, 13, and 14). As shown above, the total VOC content for the selected test pit samples ranged from 3,583 ug/kg to 1,412,000 ug/kg.

Viewed in the above described context, the above Acme Solvent Site test pit soil results are not in conflict with the groundwater results as suggested by Hickok (1985), since the total VOC content of the test pit soils were relatively high, parent compounds were widely detected in test pit samples, and DCE, when detected, was relatively high in concentration in test pit samples. Therefore, the near perfect correlation found by Hickok (1985) between total VOC content and DCE content could indicate these compounds are derived from the same source. The presence of DCE may be due to biodegradation of the parent compounds PCE and TCE.

In addition to the artifact of well placement, the distribution of VOCs in the area groundwater could additionally be influenced by intermittent and spatially variable recharge. As discussed in Section 4.4.1, the upland area east of the WRL Site receives excess groundwater recharge along the intermittent stream during precipitation and associated runoff events. The recharge noted could complicate the groundwater VOC chemistry by locally introducing relatively clean water into the dolomite aquifer. Computer modelling by E.C. Jordan (1984) indicates that such recharge could produce the VOC pattern currently observed.

A second aspect of the influence of variable recharge are waste disposal practices used at the Acme Solvent Site. The pattern of VOCs in groundwater could be related to the pattern of disposal over time and location. Waste volumes, type, and disposal location varied over the operating life of the Acme Solvent Site, and in 1986, much of the remaining waste material at the Acme Solvent Site was removed.

Figure 4-43 is a plot of total concentration of VOCs over time for selected wells north, south and east of WRL Site. In these selected wells, VOC concentrations generally decreased from the beginning of data collection in 1982 until May 1984. After May 1984, VOC concentrations increased to a maximum in January 1985 and continued to decrease to April 1988. Of these selected wells, the well with the highest VOC content has shifted downgradient from B12 to B13. These trends are consistent with the interpretation of a passing slug of groundwater with higher VOC concentrations. Samples from piezometers B16A, B10A and B11A did not show this effect indicating that this trend is not due to bias associated with variation in sampling or analytical procedures.

<u>4.7.2 VOC Migration in Landfill Gas</u>. A previous study found landfill gas to be migrating off-site to the north, east and south of the WRL Site, but observed PCE and TCE concentrations in the groundwater to the north and east of the WRL Site are much lower than the observed concentrations in the groundwater to the southeast of the WRL Site. This PCE and TCE distribution in the groundwater does not appear to be consistent with a general release of landfill gas to the north, east, and south. In addition, the observed PCE and TCE concentrations in samples from the well nests G109/G109A

and G113/G113A indicate greater PCE and TCE concentrations with increasing depth and distance from the landfill, which is also not consistent with the WRL Site gas as a source of these compounds. Also, a previous study found gas migration only in highly permeable unsaturated soils above the bedrock, indicating that the gas was not in contact with the water table present in bedrock (Warzyn, 1980).

A gas collection system has been in operation since 1980 controlling landfill gas. Nine gas extraction wells were installed in 1980. This system was replaced with 70 wells in 1984. An additional 21 wells were installed to expand the gas extraction system around 1988. The gas collection system operates 24 hours a day, all year long, with the exception of maintenance downtime and five holidays. Methane gas was detected during the boring for well G108 to the north of the WRL Site in November 1984. The methane was detected only in the sand and gravel above the bedrock at 10 to 30 % of the Lower Explosive Limit (LEL). (The LEL for methane by volume is 5% methane in air).

There is no documentation available at the present time that WRL Site accepted significant volumes of chlorinated solvent-containing waste streams. Available data indicates the WRL Site generally accepted municipal waste with limited quantities of Illinois Special Wastes (WRS, 1984). This is consistent with the leachate data as parent compounds PCE and TCE were only sparsely detected at low levels in leachate samples; therefore, it is unlikely that significant amounts of these chlorinated ethenes could be or were in the WRL Site gas.

The WRL leachate VOC composition is in sharp contrast to the groundwater samples from well G113A. Samples from well G113A contained PCE as high as 80 ug/L (Round 2) and TCE as high as 160 ug/L (Round 2). The WRL leachate generally contained much higher levels of toluene, ethylbenzene, and xylenes than chlorinated ethenes, but the groundwater samples from wells (G113, G113A) in the southeast corner exhibited the opposite trend where chlorinated ethenes were detected at much higher concentrations than toluene, ethylbenzene, and xylenes. This does not support that the WRL gas is a source of parent compounds PCE and TCE in the southeast corner.

Assuming simple equilibrium between landfill gas and leachate, the partial pressure of an individual constituent in the landfill gas can be calculated using Henry's law as follows:

$P_a = H_l R T_l C_l$ where:

- P_a = partial pressure of constituent A
- H_l = dimensionless Henry's law constant for constituent A at the leachate temperature
- R = gas constant (0.08206 atmospheres-liter/mole ^{0}Kelvin)
- T_l = leachate temperature (approximately 25^{0}C for samples collected at the WRL Site)
- C_l = concentration of constituent A in leachate

Assuming the worst case scenario of negligible diffusion or dispersion, and a continuous source of landfill gas coming in contact with groundwater present in the saturated zone, the Henry's law relationship between the landfill gas and groundwater at equilibrium would be calculated as follows:

$P_a = H_g R T_g C_g$ where:

- P_a = partial pressure of constituent A based on its concentration in the landfill leachate
- H_g = dimensionless Henry's law constant for constituent A at the groundwater temperature
- R = gas constant
- T_g = groundwater temperature (approximately 10 ^{0}C for samples collected at the WRL Site)
- C_g = concentration of constituent A in groundwater

Combining the two equations yields the following relationship between the concentration of constituent A in the groundwater versus the leachate under this worst case scenario:

$$C_g/C_l = (H_l/H_g)(T_l/T_g)$$

The ratios of the concentration in groundwater versus the leachate for the worst case equilibrium scenario discussed above were calculated for the compounds PCE and TCE. Henry's law constants were obtained from Hutzler, et al. (June 1989). The corresponding ratios and Henry's law constants are presented below:

Compound	H_l (25^{0}C)	H_g (10^{0}C)	C_g/C_l
trichloroethene (TCE)	0.41690	0.23154	1.95
tetrachloroethene (PCE)	0.69892	0.36410	2.08

Temperature differences between the landfill leachate and upgradient groundwater are not expected to account for the higher PCE and TCE concentrations observed in groundwater samples from the upgradient wells as compared to the leachate samples. Under the worst case equilibrium scenario presented above, PCE and TCE could theoretically be present at concentrations up to approximately two times higher in the groundwater than the leachate. The levels observed in samples from well G113A were much higher. Under more realistic conditions, factors such as landfill gas dilution with ambient soil gas, adsorption to soil, dilution of soil water with precipitation and groundwater, as well as diffusion and dispersion of landfill gas vapors, would make the resulting groundwater concentration less than values calculated using this worst case equilibrium scenario.

4.7.3 Household Septic Leaching Fields. Another potential source of local influence on the groundwater in the southeast corner which should be considered are septic leach fields, since they have been shown to have the potential to pollute the groundwater (Wehrmann, 1983). DeWalle, et al. (1985) documented the presence of chlorinated solvents such as TCE in septic tank effluent. Fathepure, et al. (1987) reported that TCE is used as a septic tank cleaning fluid. 1,4-Dichlorobenzene is a common ingredient in numerous household consumer products such as dyes, pesticides, air deodorizers, moth balls (Verschueren, 1983) and toilet bowl deodorants (Howard, 1990).

Chloride concentrations in septic effluents ranging up to 700 mg/L were reported by Wehrmann (1983) (p. 66). A significant source of chlorides to a septic system can be the use of a water softener. Typical salt usage in the softener system can be as much as 80 pounds per month, but will vary with water usage and hardness. The backflushing cycle of the system discharges a highly concentrated calcium and magnesium chloride solution to the septic system. The backflushing cycling period is variable, but may occur up to three to seven times a week.

Table 4-5

Table of Dilution Factors[a]

Round	Well	(A) Chloride Dilution	(B) Chlorinated Ethenes dilution without Background
1	B15	6.5	132
	B15R	8.7	2.4
	B15P	neg[b]	3.5
	P1	17.8	5.4
	MW106	9.8	3.1
	G116	neg[b]	undefined[b]
	G116A	171	5.3
2	B15	4.3	24.7
	B15R	12.4	2.9
	B15P	undefined[b]	4.1
	P1	26.7	2.6
	MW016	11.3	2.8
	G116	neg[b]	undefined[b]
	G116A	52.2	1.8

(a) Chloride dilution factors represent a rough estimate of the potential dilution of WRL leachate at a given well location adjusted for background. Ethene dilution factors are a similar representation of leachate dilution at a given well for ethenes without consideration of background. Because chloride is a conservative parameter, and ethenes are a non-conservative parameter, the ethene dilution factor is expected to be greater than or equal to the chloride dilution factor if WRL leachate were the source of ethenes.

(b) A negative indicates that it is mathematically impossible to dilute the leachate to a concentration lower than the given background value (i.e. background (15 mg/l chloride) is greater than the downgradient well result). Undefined means the denominator is zero, and occurs when background equals the downgradient well result or when no chlorinated ethenes were in the downgradient well sample.

$$A = \frac{\text{(Concentration in Well) - (Average Leachate Concentration (4,480 mg/l Cl)}}{\text{(Background Concentration) - (Concentration Measured in Well)}}$$

$$B = \frac{\text{(Concentration in Well) - (Average Leachate Concentration (77.2 ug/l VOC)}}{\text{- (Concentration in Well)}}$$

Note:
Leachate averages based upon Rounds 1 through 4 chloride and chlorinated ethene (VC, 1,2-DCE, TCE and PCE) data.

TABLE 4-7

Validated Ambient Air Volatiles Results

Compound	Number Detects	RESULTS (mg/m^3) Minimum	Maximum	ACGIH-TLVs (mg/m^3) TWA	STEL
Carbon Tetrachloride(A2)	6	9.69x10^{-5}	1.99x10^{-4}	31	--
Chloroform (A2)	5	2.5x10^{-5}	7.53x10^{-5}	49	--
Hexane	2	3.25x10^{-3}	5.97x10^{-2}	176	--
1,1-Dichloroethane	1	--	1.2x10^{-4}	810	1010
1,2-Dichloropropane	1	--	1.78x10^{-3}	347	509
Ethylbenzene	6	1.28x10^{-4}	2.11x10^{-3}	434	543
p-Dichlorobenzene	6	3.09x10^{-5}	2.59x10^{-4}	451	661
Methylene Chloride (A2)	6	1.47x10^{-2}	3.89x10^{-2}	174	--
Styrene	5	5.72x10^{-5}	1.73x10^{-4}	213	426
Tetrachloroethene	6	2.74x10^{-5}	1.08x10^{-4}	339	1368
Toluene	5	8.38x10^{-4}	1.36x10^{-2}	377	565
Trichloroethene	6	3.46x10^{-5}	2.04x10^{-5}	269	1070
O-Xylene	6	1.57x10^{-4}	1.18x10^{-3}	434	651
M+P-Xylene	4	9.21x10^{-4}	3.16x10^{-3}	434	651
Isopropyl Benzene	5	1.26x10^{-5}	4.65x10^{-4}	246	--

Total
Maximum = 0.122 mg/m^3

NAAQS - Hydrocarbons (non-methane) 0.16 mg/m^3

(A2) - Suspected human carcinogen

Table 4-1

Results of Phase I Leachate Analyses for Volatile Organics by GC/MS (ug/L) and Jordan (1984) Leachate Results

	LLMHW 4/8/88 Round 1	LLG7 6/16/88 Round 2	LLD1 6/8/89 Round 4	LLD1-DUP 6/8/89 Round 4	LLF7 6/8/89 Round 4	LLG4 6/8/89 Round 4	LLMHE 6/8/89 Round 4	Leach 1 5/84 (Jordan, 1984)	Leach 2 5/84 (Jordan, 1984)
Vinyl Chloride	12	90	54	55	12	—	3J	—	—
Total 1,2-Dichloroethene	4.3	24	21	22	—	220D	6	0.9	15
Tetrachloroethene	—	—	—	—	—	17	—	—	—
Benzene	—	6.2B	—	—	—	—	—	—	4.4
Toluene	67B	270B	320JE	310JE	47J	150J	18J	5.4	260
Ethyl Benzene	21	56B	49J	49J	65J	77J	—	—	16
Xylenes	89	130B	190JX	170JX	230JX	300JX	80JX	—	76
Chloroform	16	—	—	—	—	—	—	—	—
Chloromethane	—	—	2J	3J	—	—	—	—	—
Chloroethane	—	—	10	11	—	—	—	—	—
Methylene Chloride	—	—	13B	10B	—	—	—	—	44
Acetone	—	—	14000JBD	18000JBD	76J	—	—	—	—
1,1-dichloroethane	—	—	53	57	—	—	—	—	7.0
2-butanone	—	—	22000JD	20000JD	26J	22JD	—	—	—
1,2-dichloropropane	—	—	3J	3J	—	—	—	—	—
4-methyl-2-pentanone	—	—	1,600DJ	1,400DJ	—	—	—	—	—
2-hexanone	—	—	240JE	260JE	—	65	—	—	—

Notes:

B = Also noted in laboratory method blank
J = Estimated value
— = Not detected
D = Diluted sample
E = Exceeded calibration range
X = Manually quantified
A Round 3 GC/MS confirmation samples was not collected.

Table 4-2

Results of Phase II Leachate Analyses for Volatile Organics by GC/MS (ug/l)

	LLMHW Round 5	LLN8 Round 5	LLMHE Round 5	LL2-Dup Round 5	LLL2 Round 5	LLE3 Round 5	LLD1 Round 5
Vinyl Chloride	--	--	2J	3J	3J	270D	31
Total 1,2-Dichloroethene	--	--	5	19	19	2100D	11
Tetrachloroethene	--	--	--	--	--	--	0.7J
Benzene	3J	--	2J	3J	3J	3J	4J
Toluene	100B	120B	25B	190B	190B	99B	380JD
Ethyl Benzene	49	38	5	29	29	59	49
Xylenes	160X	110X	98X	69X	70X	180X	170X
Chloroform	--	--	--	--	--	--	--
Chlrormethane	--	--	--	--	--	--	15
Chloroethane	--	--	--	--	--	--	9J
Methylene Chloride	--	--	--	--	--	--	100JB
Acetone	--	--	--	--	--	--	--
1,1-dichloroethane	--	--	--	--	--	--	55
1,1-dichloroethene	--	--	--	--	--	1J	--
2-butanone	--	--	--	--	--	--	7900JD
1,2-dichloropropane	--	--	--	--	--	34	3J
4-methyl-2-pentanone	43J	--	--	--	--	--	--
Chlorobenzene	--	--	3J	--	--	4J	2J
Styrene	--	--	--	4J	4J	7	--
Trichloroethene	--	--	--	--	--	29	1J
Carbon Disulfide	--	--	--	--	--	--	1J
2-Hexanone	--	--	--	--	--	--	39JX

Notes:

B = Also noted in laboratory method blank.
J = Estimated value.
-- = Not detected.
D = Diluted sample.
E = Exceeded calibration range.
X = Manually quantified

V160R09JAH/CWB/BOF/MJH

Table 4-3

Results of Leachate Analyses for Semi-Volatiles

Compound	WRL Observed Range of Detections (ug/L)	Number of Detections*
Phenol	140	1
1,3-Dichlorobenzene	19	1
1,4-Dichlorobenzene	22-27	2
2-Methylphenol	27-85	3
4-Methylphenol	30-55	2
2,4-Dimethylphenol	33-160	6
Benzoic Acid	1200	1
Napthalene	6-50	6
2-Methylnapthalene	8-23	2
Fluorene	17	1
Phenanthrene	6-53	3
Anthracene	2	1
Fluoranthene	12-22	2
Pyrene	9	1
bis(2-Ethylhexyl)phthalate	80-1200	6
Dibenzofuran	11	1
Di-n-octyl phthalate	13-170	4

* Total number of detections reported out of 11 samples analyzed in Round 1 and Round 5.

Table 4-4

Comparison of Results of Leachate Analyses with Reported Values for Sanitary Landfill Leachate

Parameter	WRL Observed Range	Typical Values*
pH (standard units)	6.63 - 7.95	5 - 8
Specific conductance (umhos/cm)	6,520 - >50,000	NR
Alkalinity (mg/L)	2,600 - 14,400	1,000 - 10,000
Chloride (mg/L)	1,160 - 17,300	100 - 3,000
Sodium (mg/L)	968 - 10,200	200 - 2,000
Calcium (mg/L)	29.9 - 241	200 - 3,000
Magnesium (mg/L)	30.8 - 812	50 - 1,500
Potassium (mg/L)	608 - 1,750	200 - 2,000
Iron (mg/L)	5.47 - 263	50 - 600
Lead (ug/L)	26 - 1,450	20 - 300
Copper (ug/L)	ND - 5,720	45 - 300
Zinc (ug/L)	191 - 15,400	28,000 - 30,000
Chromium (ug/L)	143 - 933	120
Cadmium (ug/L)	ND - 226	250
Arsenic (ug/L)	8 - 318	110 - 160
Nickel (ug/L)	323 - 1,130	600 - 1,050
Barium (ug/L)	78 - 4,710	NR
Selenium (ug/L)	ND-12	NR
Mercury (ug/L)	ND - 5.9	NR
Cyanide (mg/L)	0.04 - 6.0	NR

* after Cope, et al., 1983
ND not detected
NR not reported

Toluene was not detected in Round 1. Toluene was detected at concentrations ranging from 0.24 ug/L to 3 ug/L in Round 2 groundwater samples from four wells: B4, B15, G110, and G114. Toluene was not detected in the wells sampled in Rounds 3 and 4, which included G110 and G114. Wells B4 and B15 were not sampled in Rounds 3 or 4.

Total xylenes were observed in Rounds 1 and 2 groundwater samples from ten wells: B4, B13, B15, B15R, G109A, G110, G113, G113A, G114, and MW106. Of the wells sampled in Rounds 3 and 4, total xylenes were not detected in samples from well B13, G109A, or G114. Total xylenes were detected only in the Round 3 sample from well B15R (1JX ug/L) and well G110 (13 X ug/L), and the Round 4 sample from well G110 (50 X ug/L)(X means the concentration is manually derived). The total xylenes concentration observed at well G110 have consistently increased from Round 1 (not detected) to Round 4 (50X ug/L).

<u>Other VOCs</u>: The compound 1,2-dichloropropane was detected in Rounds 1 and 2 groundwater samples from 18 wells: B4, B10, B12, B13, B15, B15R, G108, G109, G109A, G110, G113, G113A, G114, G115, MW106, P1, P3R, and P4R. Of the wells sampled in Rounds 3 and 4, the presence of 1,2-dichloropropane was consistently confirmed in samples from wells B13, G109A, G110, and G114. The compound 1,2-dichloropropane was not detected in Rounds 3 or 4 samples from wells G115, P1, P3R and P4R. 1,2-Dichloropropane was not detected in the Round 3 sample from well B15R, but was detected in Round 4 (2J ug/L), and similarly was detected in the Round 3 sample from well G109 (3J ug/L), but not detected in Round 4.

Based upon Rounds 1 and 2 groundwater samples, chlorobenzene was detected in samples from 12 wells: B12, B13, B15, B15R, G109, G109A, G110, G113A, G114, MW106, P1, and P4R. Of the wells sampled in Rounds 3 and 4, the presence of chlorobenzene was again detected in samples from wells B13, G109A, G110, G114, and B15R, but not detected in the samples from well P4R. Chlorobenzene was detected in the Round 3 sample from well G109 (2J ug/L), but not in the Round 4 sample. In addition, chlorobenzene was detected only in one of the four sampling rounds at well G115 (Round 3).

Table 4-6

The Ratio of Chlorinated Ethenes to Total Chlorinated Ethenes (as a Percent) in Selected Groundwater Samples Round 2

	PCE	TCE	DCE	VC
Dolomite Wells East of Lindenwood Road				
B4	42.3	18.3	39.3	0.1
B16	12.0	14.3	73.7	0.0
B16A	5.0	5.5	89.5	0.0
B11A	34.1	15.9	50.0	0.0
G108	32.5	23.1	43.1	1.3
Dolomite Wells West of Lindenwood Road				
B12	5.7	6.6	80.2	7.5
G109	18.3	5.9	55.9	19.9
G109A	6.6	26.1	59.1	8.3
G113	22.1	4.6	63.4	10.0
G113A	14.1	30.0	53.4	2.4
G111	16.6	11.6	71.8	0.0
G114	0.0	7.4	42.4	50.2
G110	2.0	7.6	13.0	77.4
B13	8.2	15.3	72.0	4.5
P6	19.4	22.5	58.1	0.0
Unconsolidated Material Wells				
B15R	8.1	23.2	52.6	16.1
G115	0.0	0.0	29.2	70.8
P3R	0.0	13.2	71.2	15.7
P4R	15.7	15.7	66.5	2.0
MW106	0.0	32.8	52.0	15.2
P1	0.0	16.7	62.9	20.4
G116	0.0	0.0	0.0	0.0
G116A	19.6	14.9	65.5	0.0

QUALITY CONTROL
Drafting Standards
Lead Professional
Section
PM
Division
Other

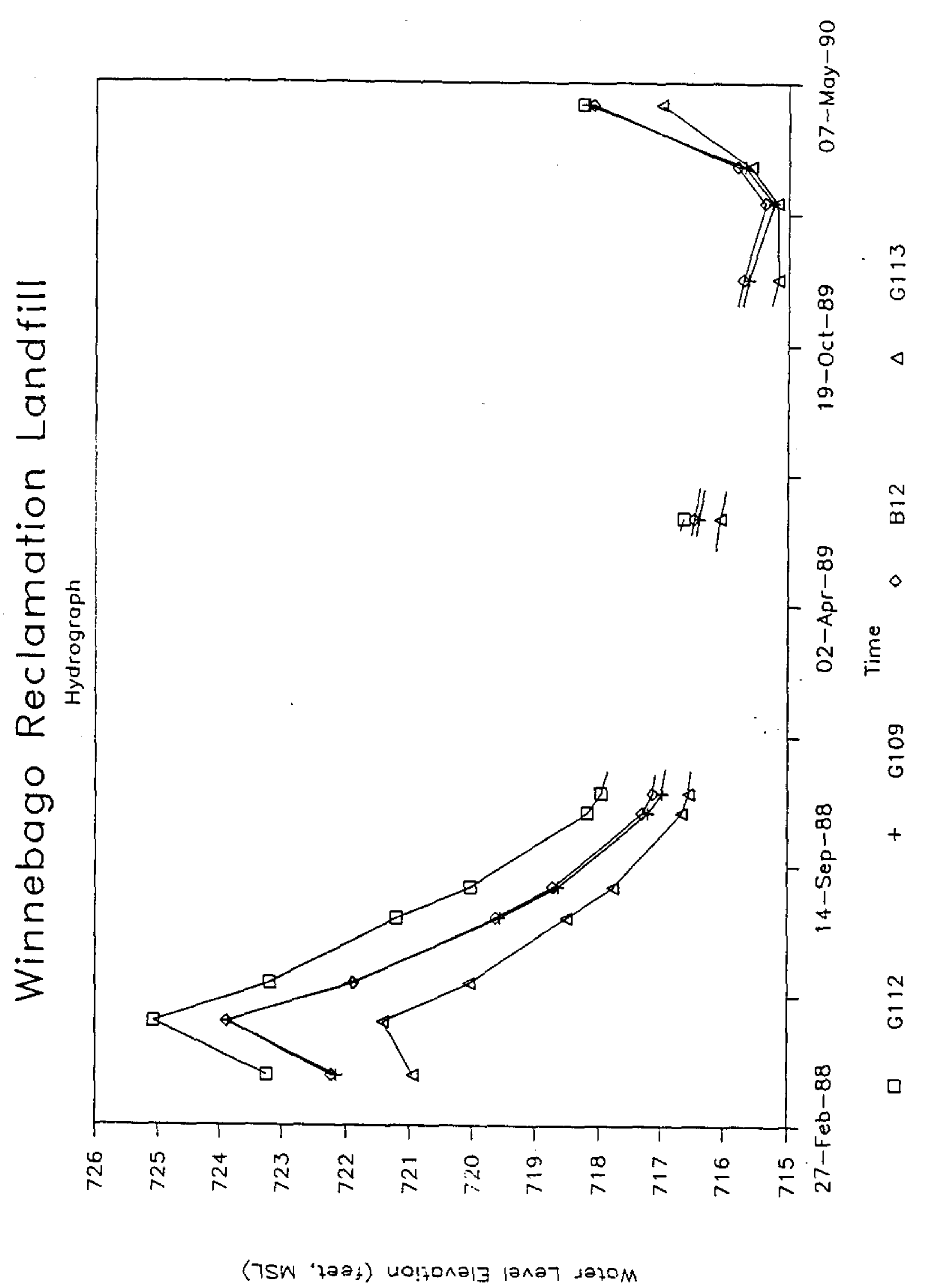
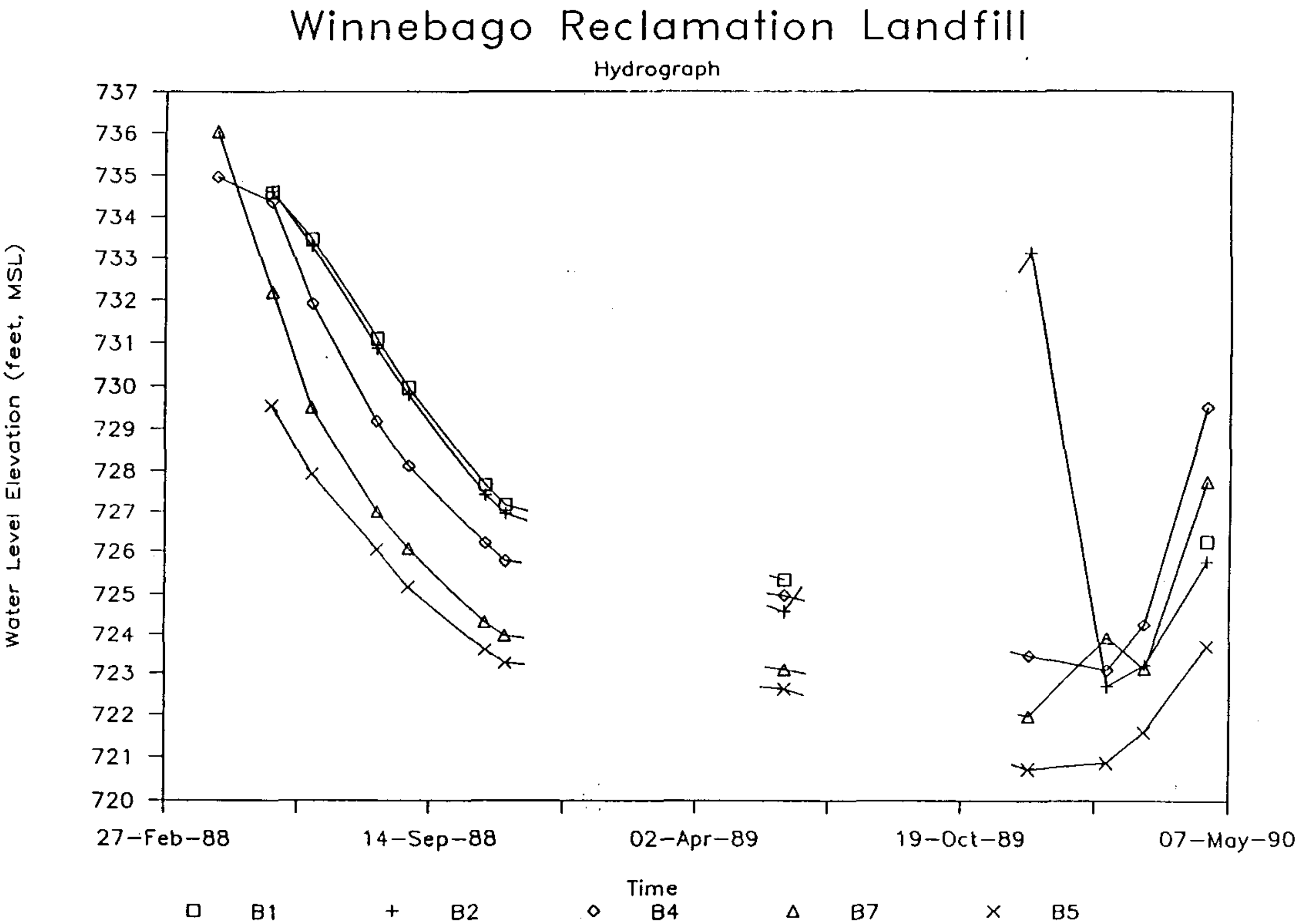


FIGURE 4-14

WARZYN	HYDROGRAPH FOR WELLS G112, G109, B12, AND G113	Drawn DLL	Checked MJH	App'd Gary E. Parker
	WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS			Date 4-01-91
				13160 A27

QUALITY CONTROL

	INITIAL	DATE		INITIAL	DATE
Drafting Standards			PM		
Lead Professional			Division		
Section			Other		



WARZYN

HYDROGRAPH FOR WELLS B1, B2, B4, B7, AND B5
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

Drawn DLL

Checked MJH

App'd [signature]

Date 4-01-91

13160

AR

FIGURE 4-15

QUALITY CONTROL

	INITIAL	DATE		INITIAL	DATE
Drafting Standards			PM		
Lead Professional			Division		
Section			Other		

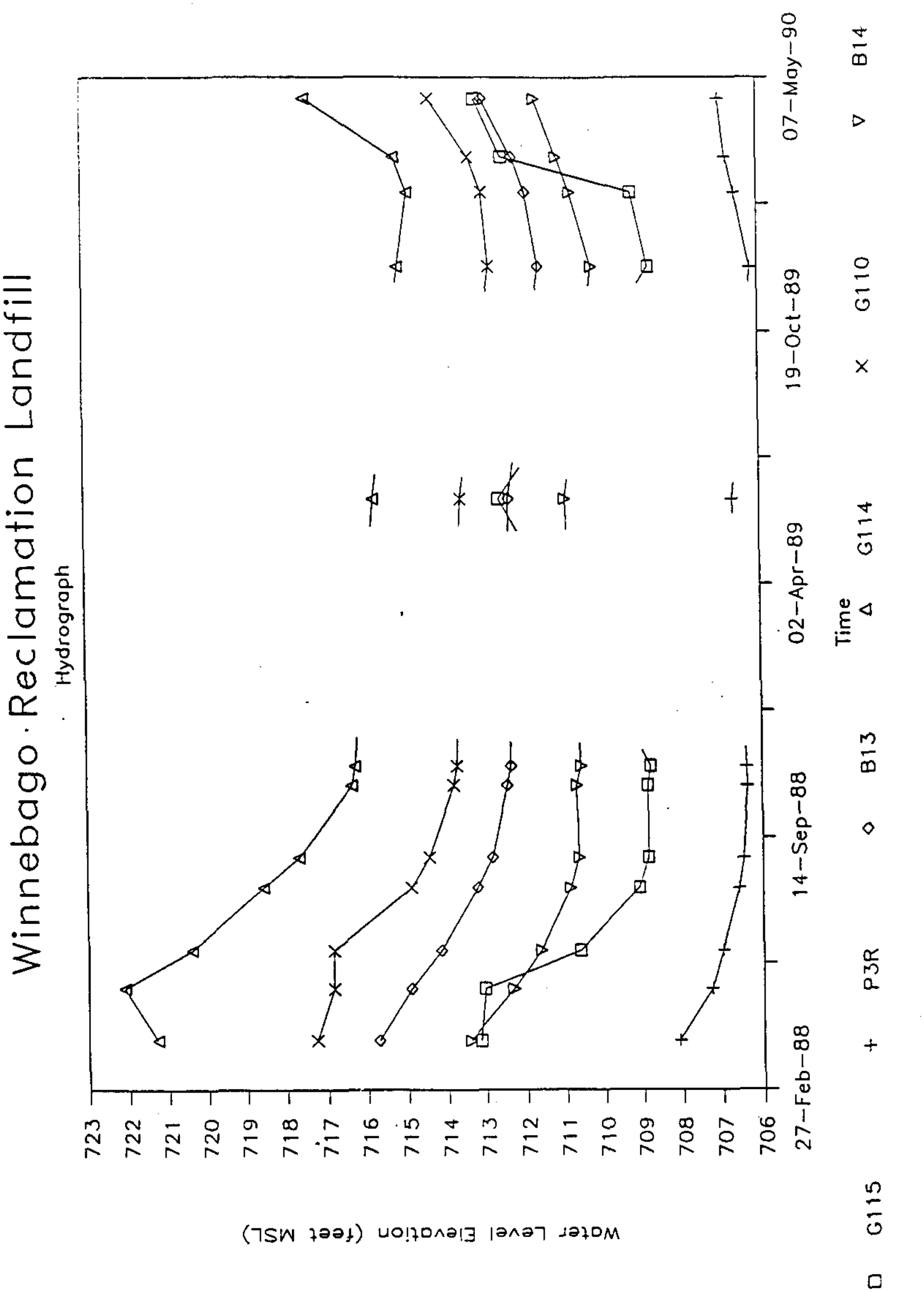
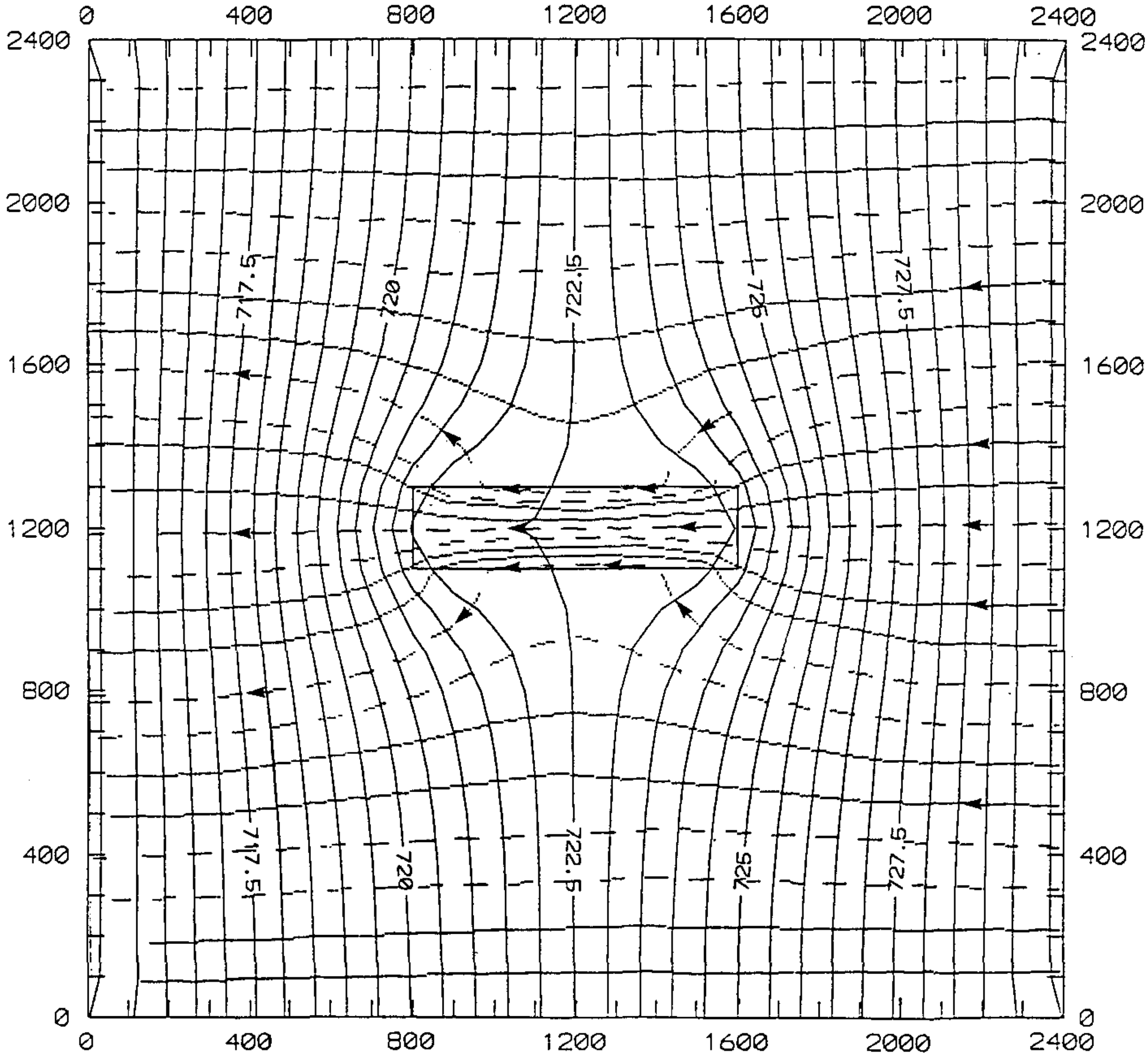


FIGURE 4-16

WARZYN	HYDROGRAPH FOR WELLS G115, P3R, B13, G114, G110, AND B14	Drawn DLL	Checked MJH
	WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS	App'd Gary E. Parker	Date 4-01-91
		13160	A0



LEGEND

HIGH PERMEABILITY ZONE, 100 TIMES GREATER THAN SURROUNDING AQUIFER

—717— POTENTIOMETRIC CONTOUR LINE

GROUNDWATER FLOW LINE INDICATING FLOW DIRECTION (I.E. FLOW LINES ARE PERPENDICULAR TO POTENTIOMETRIC CONTOUR LINES)

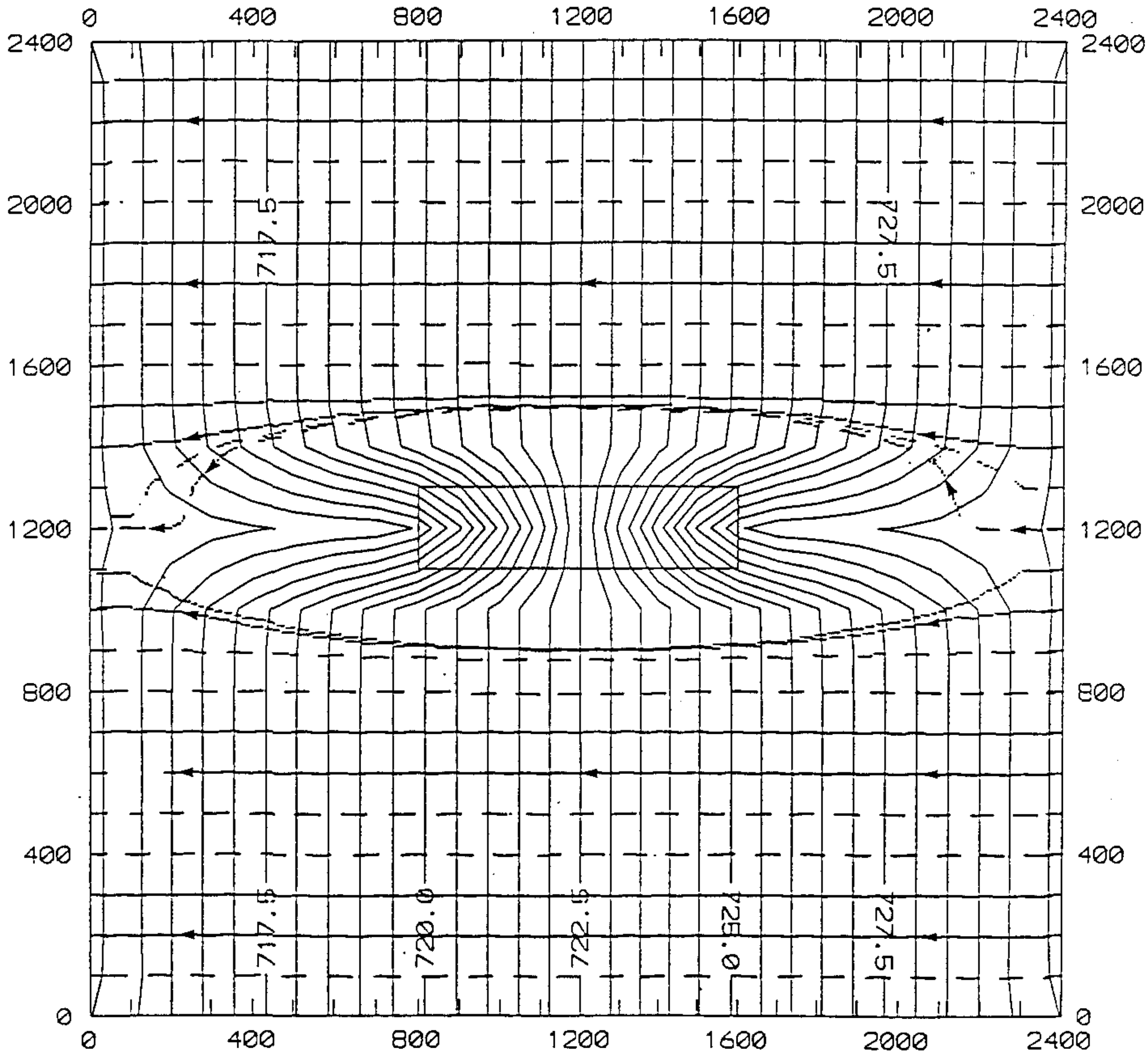
NOTES

1. FLOW CONFIGURATION VALID IN PLAN VIEW OR CROSS-SECTIONAL VIEW.

FIGURE 4-17

WARZYN	HYPOTHETICAL HIGH PERMEABILITY ZONE WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS	Drawn DLL Revisions	Checked MJH	App'd [signature] Date 4-01-91 13160 **A10**

Drafting Standards / Lead Professional / Section — QUAL INITIALS / CONTR DATE / INITIALS DATE — PM / Division / Other



LEGEND

LOW PERMEABILITY ZONE, 100 TIMES LESS THAN SURROUNDING AQUIFER

—717— POTENTIOMETRIC CONTOUR LINE

GROUNDWATER FLOW LINE INDICATING FLOW DIRECTION (I.E. FLOW LINES ARE PERPENDICULAR TO POTENTIOMETRIC CONTOUR LINES)

NOTES

1. FLOW CONFIGURATION VALID IN PLAN VIEW OR CROSS-SECTIONAL VIEW.

FIGURE 4-18

WARZYN	HYPOTHETICAL LOW PERMEABILITY ZONE WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS	Drawn DLL Revisions	Checked MJH	App'd [signature] Date 4-01-91 12160 A11

QUALITY CONTROL INITIALS DATE ...g St... ...s Lead Professional Section Division Other

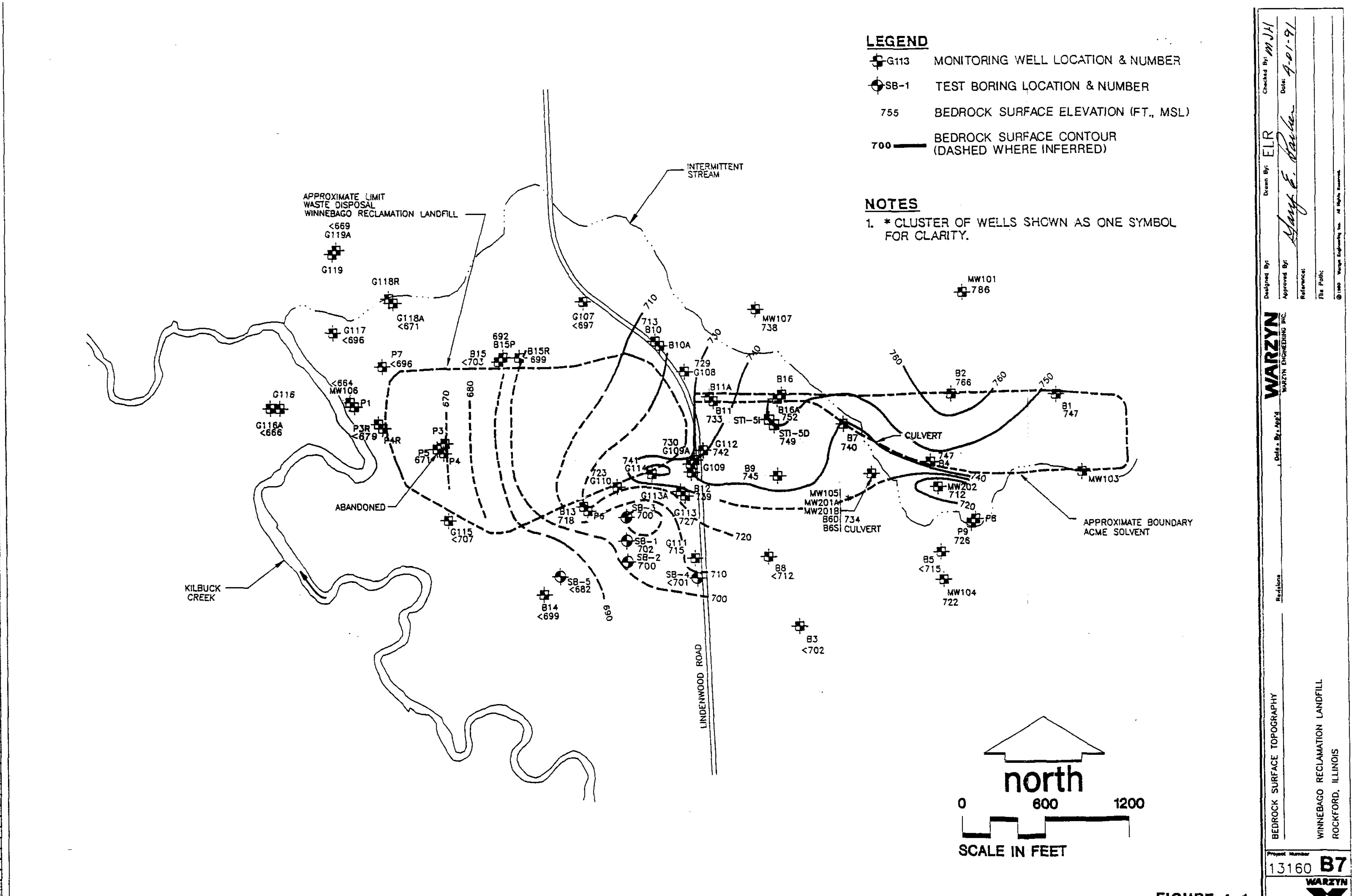

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
SB-1 TEST BORING LOCATION & NUMBER
755 BEDROCK SURFACE ELEVATION (FT., MSL)
700 BEDROCK SURFACE CONTOUR (DASHED WHERE INFERRED)
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
INTERMITTENT STREAM
APPROXIMATE LIMIT WASTE DISPOSAL WINNEBAGO RECLAMATION LANDFILL
ABANDONED
KILBUCK CREEK
LINDENWOOD ROAD
CULVERT
APPROXIMATE BOUNDARY ACME SOLVENT
G119A <669
G119
G118R
G118A <671
G117 <696
G116
G116A <666
MW106 <664
P1
P3R <679
P4R
P7 <696
B15 <703
692 B15P
B15R 699
P3
P5 671
P4
G107 <697
713 B10
B10A
729 G108
B11A
B11 733
STI-5I
STI-5D 749
B16
B16A 752
MW107 738
MW101 786
B2 766
B1 747
B7 740
B4 747
MW202 712
MW103
P8
P9 726
B5 <715
MW104 722
B3 <702
B8 <712
B9 745
MW105
MW201A
MW201B
B6D 734
B6S
730 G109A
G112 742
G109
741 G114
723 G110
B12 739
G113A
G113 727
SB-3 700
SB-1 702
SB-2 700
G111 715
SB-4 <701
B13 718
P6
G115 <707
SB-5 <682
B14 <699
670
680
690
700
710
720
730
740
750
760
north
0 600 1200
SCALE IN FEET
BEDROCK SURFACE TOPOGRAPHY
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
WARZYN
Drawn By: ELR
Checked By: MJH
Date: 4-01-91
Project Number 13160 B7

FIGURE 4-1

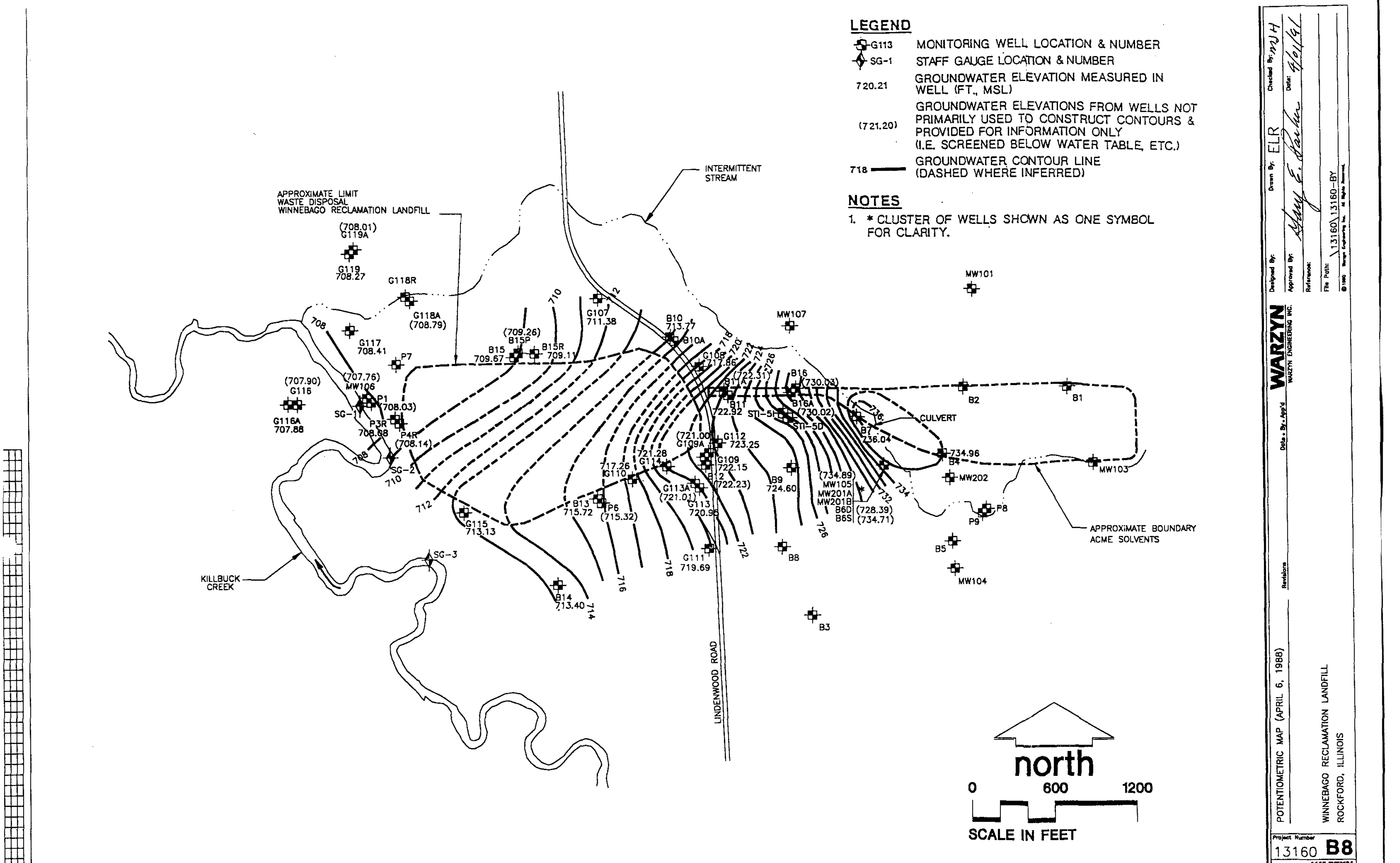
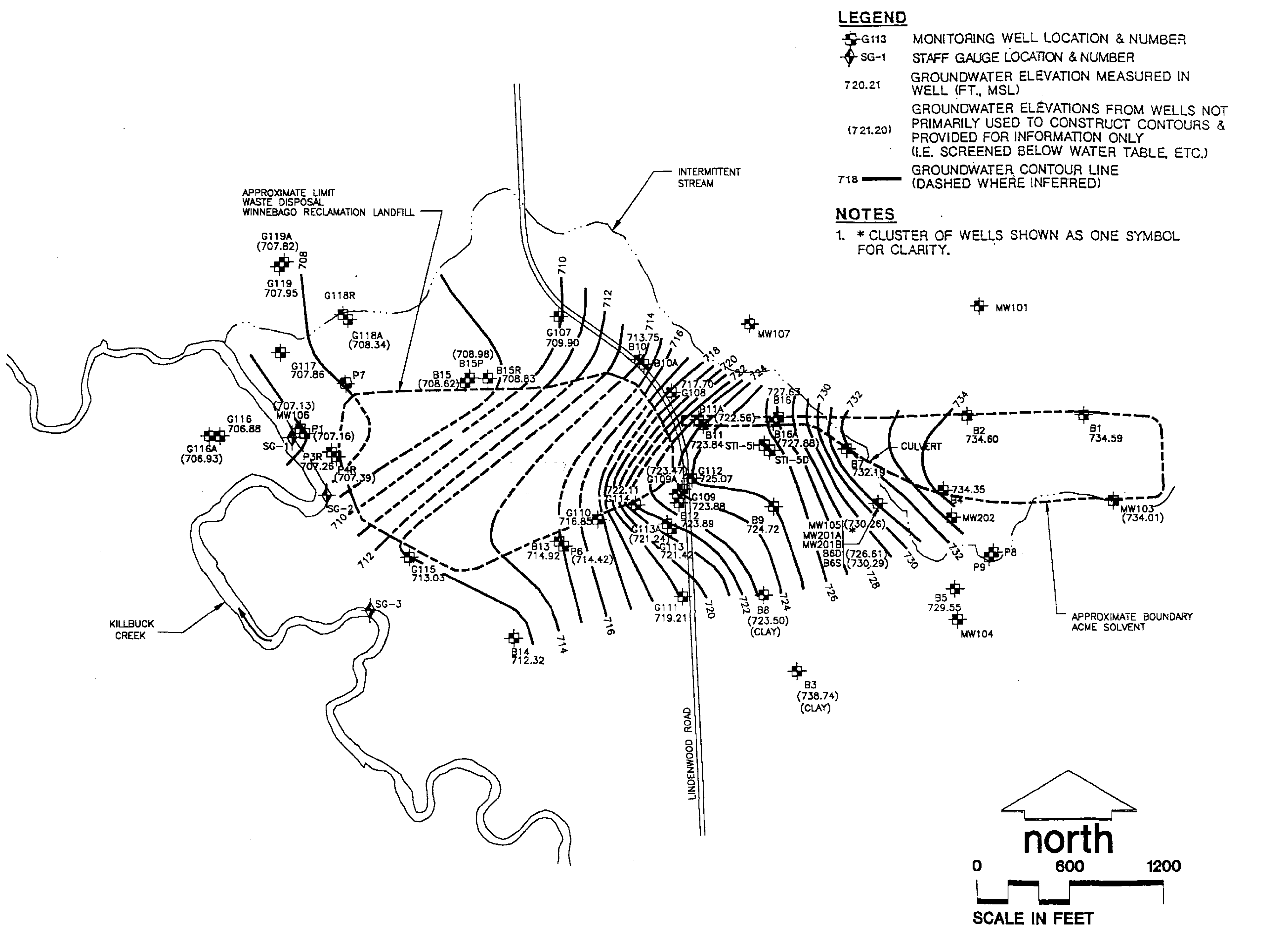

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
SG-1 STAFF GAUGE LOCATION & NUMBER
720.21 GROUNDWATER ELEVATION MEASURED IN WELL (FT., MSL)
(721.20) GROUNDWATER ELEVATIONS FROM WELLS NOT PRIMARILY USED TO CONSTRUCT CONTOURS & PROVIDED FOR INFORMATION ONLY (I.E. SCREENED BELOW WATER TABLE, ETC.)
718 GROUNDWATER CONTOUR LINE (DASHED WHERE INFERRED)
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
INTERMITTENT STREAM
APPROXIMATE LIMIT WASTE DISPOSAL WINNEBAGO RECLAMATION LANDFILL
KILLBUCK CREEK
LINDENWOOD ROAD
CULVERT
APPROXIMATE BOUNDARY ACME SOLVENTS
north
0 600 1200
SCALE IN FEET
POTENTIOMETRIC MAP (APRIL 6, 1988)
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
WARZYN
WARZYN ENGINEERING INC.
Drawn By: ELR
Project Number 13160 B8
File Path: \13160\13160-BY

FIGURE 4-2



POTENTIOMETRIC MAP (MAY 17, 1988)

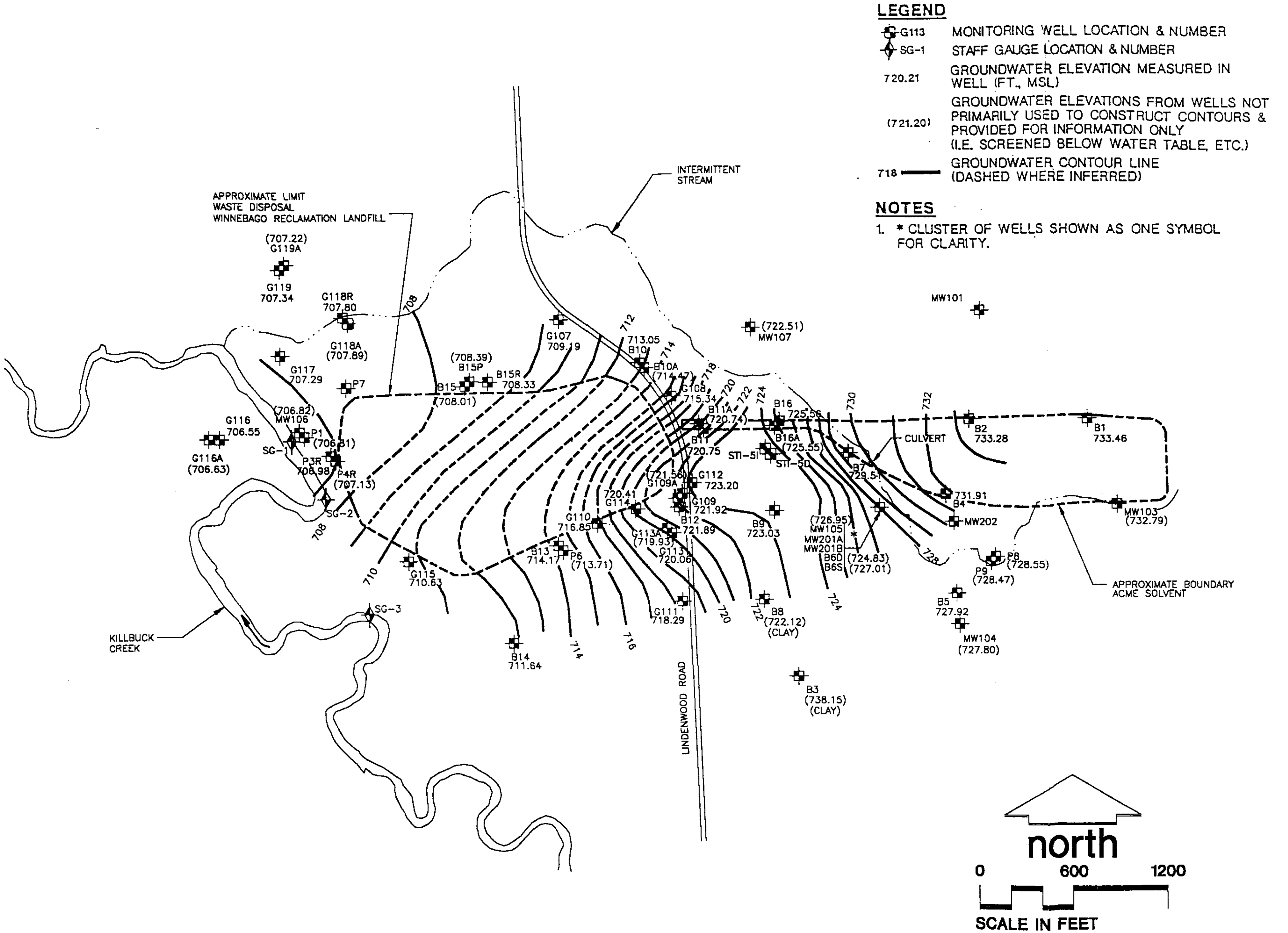
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

WARZYN
WARZYN ENGINEERING INC.

Drawn By: ELR
Checked By: mJH
Date: 9/01/91
File Path: 13160\13160-BX

Project Number 13160 B9

FIGURE 4-3



POTENTIOMETIC MAP (JUNE 16, 1988)

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

Designed By: Drawn By: ELR Checked By: Approved By: Date: 4/01/91

WARZYN ENGINEERING INC.

File Path: \13160\13160-BW

Project Number 13160B10

WARZYN

FIGURE 4-4

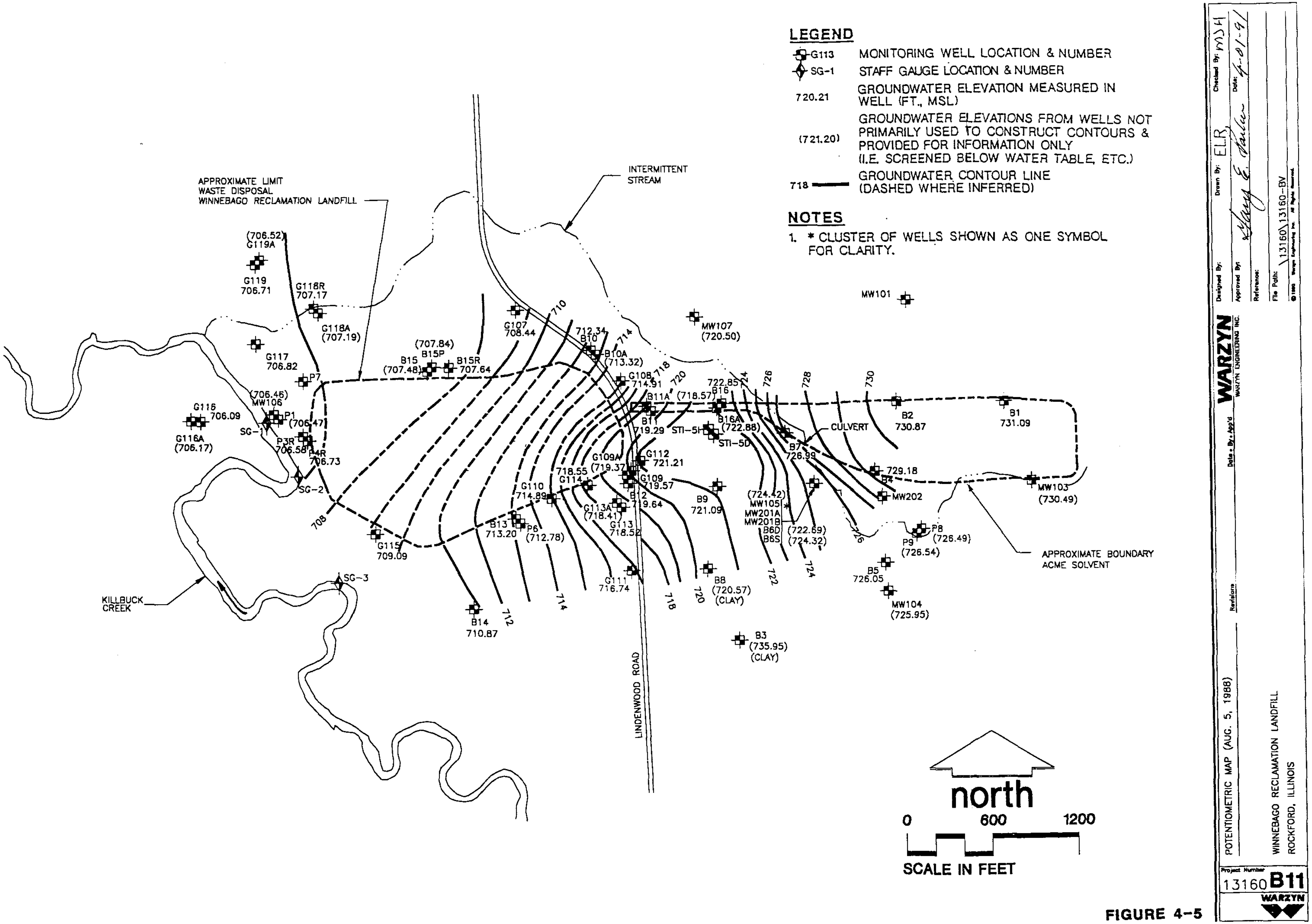
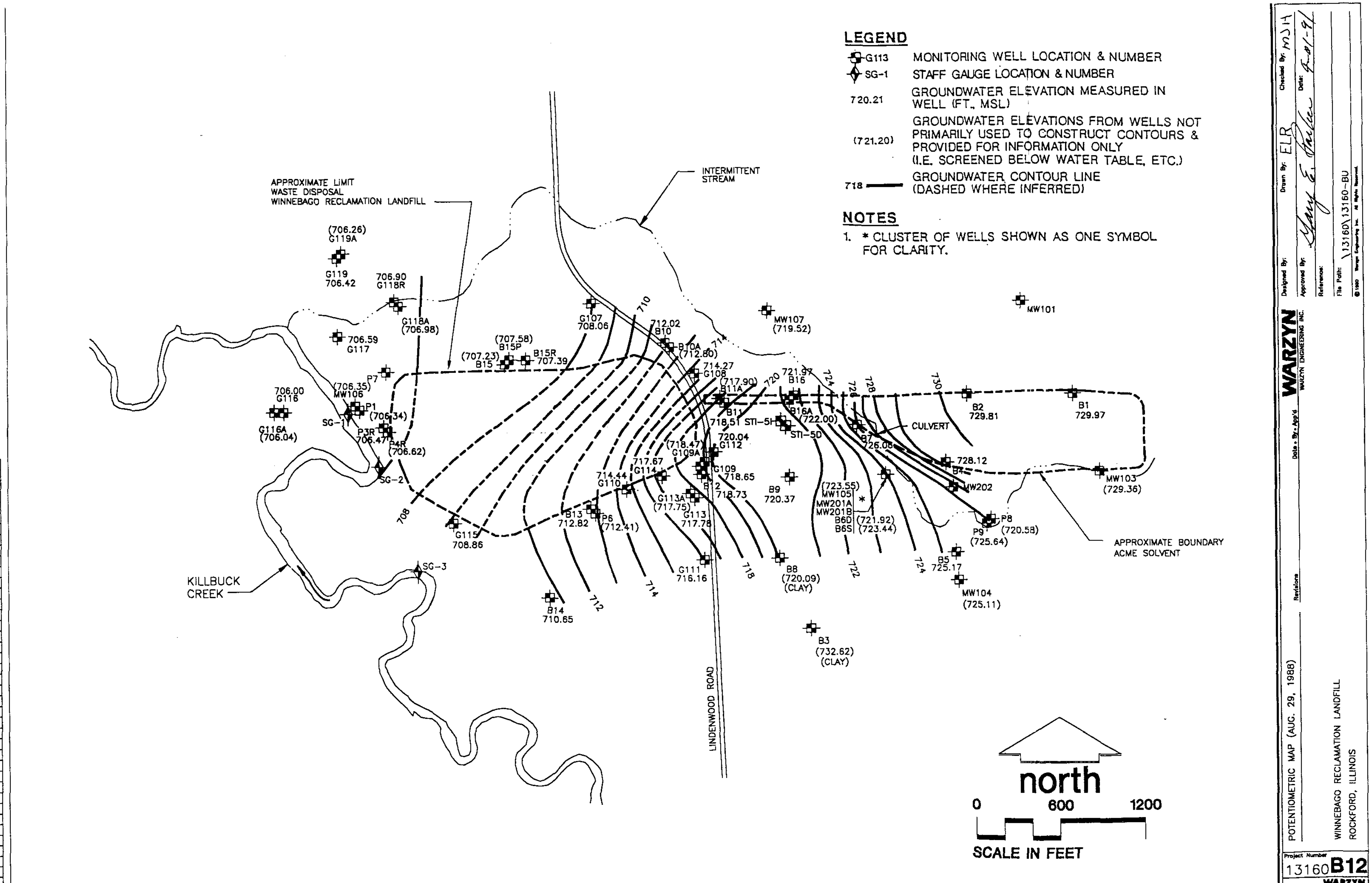

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
SG-1 STAFF GAUGE LOCATION & NUMBER
720.21 GROUNDWATER ELEVATION MEASURED IN WELL (FT., MSL)
(721.20) GROUNDWATER ELEVATIONS FROM WELLS NOT PRIMARILY USED TO CONSTRUCT CONTOURS & PROVIDED FOR INFORMATION ONLY (I.E. SCREENED BELOW WATER TABLE, ETC.)
718 GROUNDWATER CONTOUR LINE (DASHED WHERE INFERRED)
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
APPROXIMATE LIMIT WASTE DISPOSAL WINNEBAGO RECLAMATION LANDFILL
INTERMITTENT STREAM
CULVERT
APPROXIMATE BOUNDARY ACME SOLVENT
KILLBUCK CREEK
LINDENWOOD ROAD
(706.52) G119A
G119 706.71
G118R 707.17
G118A (707.19)
G117 706.82
P7
(706.46) MW106
G116 706.09
G116A (706.17)
SG-1
P1
(706.47)
P3R 706.58
P4R 706.73
SG-2
SG-3
B15 (707.48)
(707.84) B15P
B15R 707.64
G107 708.44
712.34 B10
B10A (713.32)
G108 714.91
B11A (718.57)
B11 719.29
STI-5H
STI-5D
722.85 B16
B16A (722.88)
G112 721.21
G109A (719.37)
G109 719.57
718.55 G114
G110 714.89
B12 719.64
G113A (718.41)
G113 718.52
B13 713.20
P6 (712.78)
G115 709.09
B14 710.87
G111 716.74
B9 721.09
B8 (720.57) (CLAY)
B3 (735.95) (CLAY)
MW107 (720.50)
MW101
B2 730.87
B1 731.09
B7 726.99
729.18 B4
MW202
(724.42) MW105 MW201A MW201B B6D B6S
(722.69) (724.32)
MW103 (730.49)
P8 (726.49)
P9 (726.54)
B5 726.05
MW104 (725.95)
708 710 712 714 716 718 720 722 724 726 728 730
north
0 600 1200
SCALE IN FEET
WARZYN
POTENTIOMETRIC MAP (AUG. 5, 1988)
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
13160\13160-BV
13160 B11

FIGURE 4-5



LEGEND
G113 MONITORING WELL LOCATION & NUMBER
SG-1 STAFF GAUGE LOCATION & NUMBER
720.21 GROUNDWATER ELEVATION MEASURED IN WELL (FT., MSL)
(721.20) GROUNDWATER ELEVATIONS FROM WELLS NOT PRIMARILY USED TO CONSTRUCT CONTOURS & PROVIDED FOR INFORMATION ONLY (I.E. SCREENED BELOW WATER TABLE, ETC.)
718 GROUNDWATER CONTOUR LINE (DASHED WHERE INFERRED)
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
APPROXIMATE LIMIT WASTE DISPOSAL WINNEBAGO RECLAMATION LANDFILL
INTERMITTENT STREAM
KILLBUCK CREEK
LINDENWOOD ROAD
CULVERT
APPROXIMATE BOUNDARY ACME SOLVENT
(706.26) G119A
G119 706.42
706.90 G118R
G118A (706.98)
706.59 G117
P7
(706.35) MW106
706.00 G116
G116A (706.04)
P1 (706.34)
SG-1
P3R 706.47
P4R (706.62)
SG-2
SG-3
(707.58) B15P
(707.23) B15
B15R 707.39
G107 708.06
G115 708.86
712.02 B10
B10A (712.80)
714.27 G108
(717.90) B11A
B11 718.51
STI-5H
STI-5D
720.04 G112
(718.47) G109A
G109 718.65
B12 718.73
717.67 G114
714.44 G110
G113A (717.75)
G113 717.78
B13 712.82
P6 (712.41)
G111 716.16
B14 710.65
MW107 (719.52)
721.97 B16
B16A (722.00)
B9 720.37
B8 (720.09) (CLAY)
B3 (732.62) (CLAY)
(723.55) MW105
MW201A
MW201B
B6D (721.92)
B6S (723.44)
B7 726.06
B2 729.81
B1 729.97
MW101
728.12 B4
MW202
MW103 (729.36)
P8 (720.58)
P9 (725.64)
B5 725.17
MW104 (725.11)
708 710 712 714 716 718 720 722 724 726 728 730
north
0 600 1200
SCALE IN FEET
WARZYN
WARZYN ENGINEERING INC.
Drawn By: ELR
Checked By: MJH
Date: 8-21-91
File Path: \13160\13160-BU
POTENTIOMETRIC MAP (AUG. 29, 1988)
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
Project Number 13160B12

FIGURE 4-6

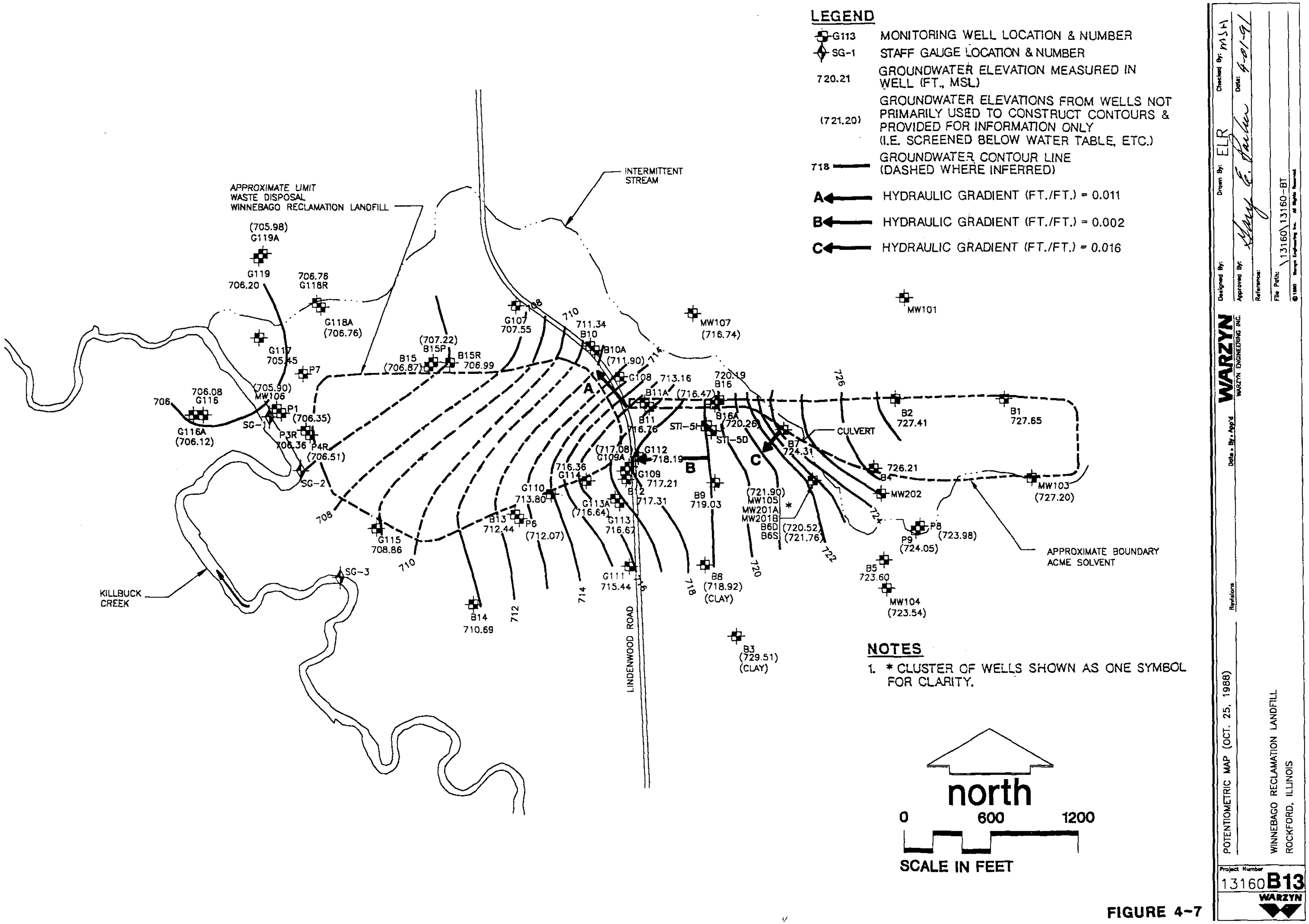
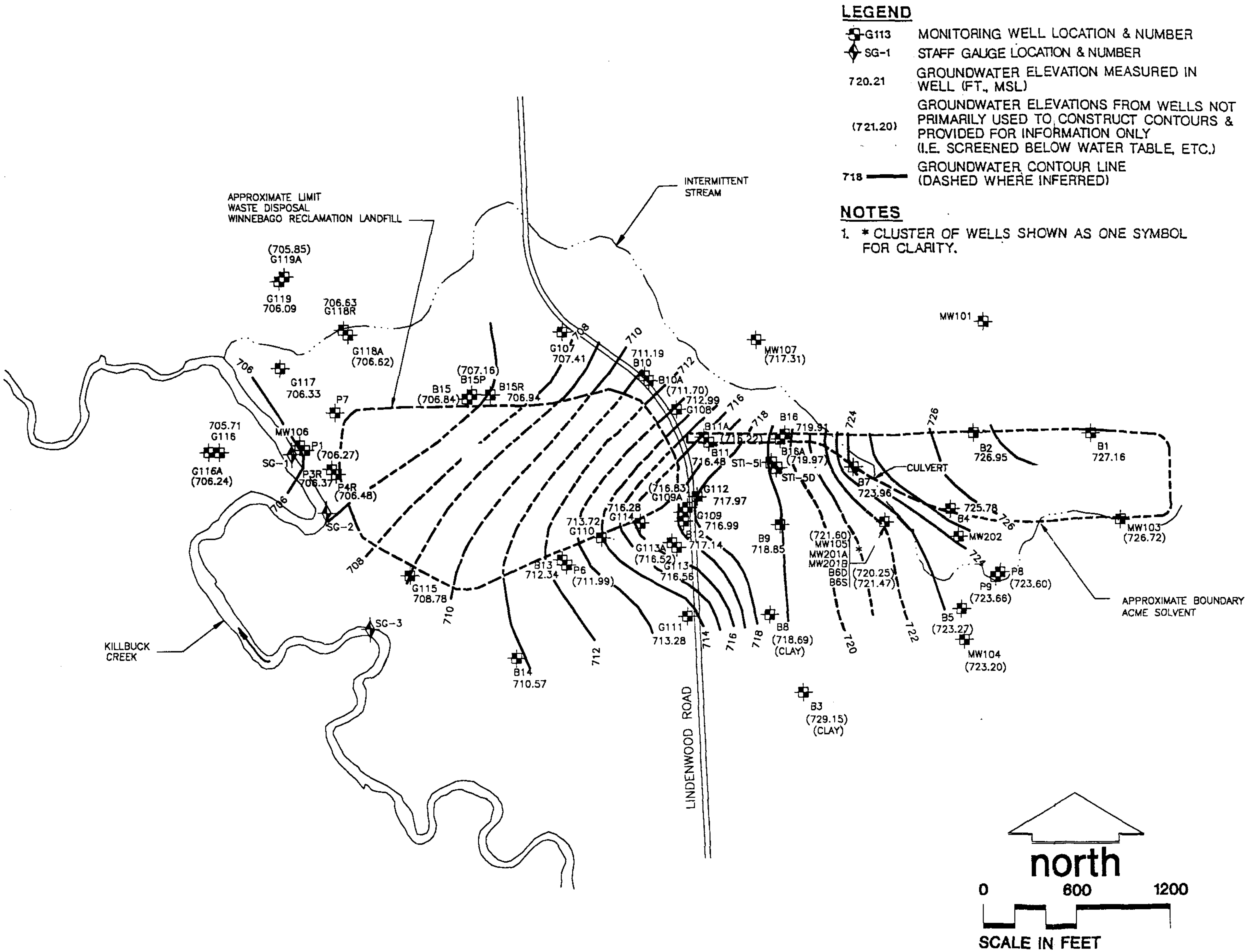

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
SG-1 STAFF GAUGE LOCATION & NUMBER
720.21 GROUNDWATER ELEVATION MEASURED IN WELL (FT., MSL)
(721.20) GROUNDWATER ELEVATIONS FROM WELLS NOT PRIMARILY USED TO CONSTRUCT CONTOURS & PROVIDED FOR INFORMATION ONLY (I.E. SCREENED BELOW WATER TABLE, ETC.)
718 GROUNDWATER CONTOUR LINE (DASHED WHERE INFERRED)
A HYDRAULIC GRADIENT (FT./FT.) = 0.011
B HYDRAULIC GRADIENT (FT./FT.) = 0.002
C HYDRAULIC GRADIENT (FT./FT.) = 0.016
APPROXIMATE LIMIT WASTE DISPOSAL WINNEBAGO RECLAMATION LANDFILL
INTERMITTENT STREAM
CULVERT
APPROXIMATE BOUNDARY ACME SOLVENT
KILLBUCK CREEK
LINDENWOOD ROAD
(705.98) G119A
G119 706.20
706.76 G118R
G118A (706.76)
G117 705.45
P7
706.08 G116
G116A (706.12)
(705.90) MW106
P1
(706.35)
SG-1
P3R 706.36
P4R (706.51)
SG-2
SG-3
G115 708.86
B14 710.69
B13 712.44
P6 (712.07)
G110 713.80
G107 707.55
B15 (706.87)
(707.22) B15P
B15R 706.99
711.34 B10
B10A (711.90)
G108 713.16
B11A (716.47)
B11 716.76
720.19 B16
B16A (720.26)
STI-5H
STI-5D
(717.08) G109A
G112 718.19
G109 717.21
716.36 G114
G113A (716.64)
G113 716.67
B12 717.31
G111 715.44
B9 719.03
B8 (718.92) (CLAY)
B3 (729.51) (CLAY)
MW107 (716.74)
MW101
(721.90) MW105 MW201A MW201B B6D B6S
(720.52) (721.76)
B7 724.31
B2 727.41
B1 727.65
726.21 B4
MW202
MW103 (727.20)
P8 (723.98)
P9 (724.05)
B5 723.60
MW104 (723.54)
706
708
710
712
714
716
718
720
722
724
726
A
B
C
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
north
0 600 1200
SCALE IN FEET
Designed By: Drawn By: ELR Checked By: MSH
Approved By: Gary E. Parker Date: 4-01-91
File Path: \13160\13160-BT
WARZYN WARZYN ENGINEERING INC.
POTENTIOMETRIC MAP (OCT. 25, 1988)
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
Project Number 13160B13
WARZYN

FIGURE 4-7



Designed By: | Drawn By: ELR | Checked By: MSH
Approved By: Gary E. Parker | Date: 9-01-91
Reference:
File Path: 13160\13160-BS
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WARZYN ENGINEERING INC.

Date • By • App'd | Revisions

POTENTIOMETRIC MAP (NOV. 9, 1988)

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

Project Number
13160B14

WARZYN

FIGURE 4-8

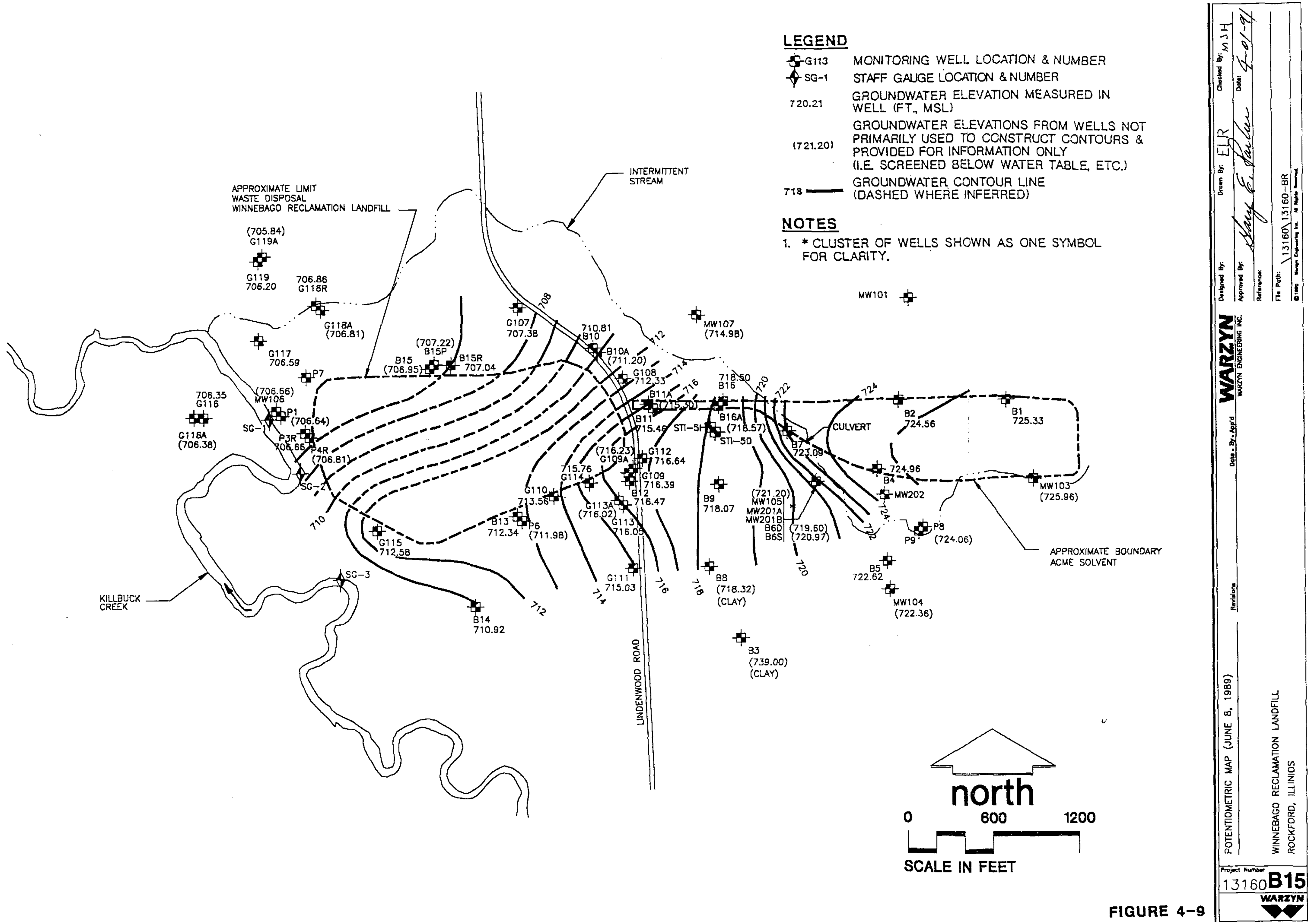

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
SG-1 STAFF GAUGE LOCATION & NUMBER
720.21 GROUNDWATER ELEVATION MEASURED IN WELL (FT., MSL)
(721.20) GROUNDWATER ELEVATIONS FROM WELLS NOT PRIMARILY USED TO CONSTRUCT CONTOURS & PROVIDED FOR INFORMATION ONLY (I.E. SCREENED BELOW WATER TABLE, ETC.)
718 GROUNDWATER CONTOUR LINE (DASHED WHERE INFERRED)
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
APPROXIMATE LIMIT WASTE DISPOSAL WINNEBAGO RECLAMATION LANDFILL
INTERMITTENT STREAM
CULVERT
APPROXIMATE BOUNDARY ACME SOLVENT
KILLBUCK CREEK
LINDENWOOD ROAD
(705.84) G119A
G119 706.20
706.86 G118R
G118A (706.81)
G117 706.59
G107 707.38
(707.22) B15P
B15 (706.95)
B15R 707.04
710.81 B10
B10A (711.20)
G108 712.33
MW107 (714.98)
MW101
706.35 G116
G116A (706.38)
(706.66) MW106
P1
SG-1
P7
P3R 706.66
P4R (706.81)
(706.64)
SG-2
SG-3
B11A (715.30)
B11 715.46
718.50 B16
B16A (718.57)
STI-5H
STI-5D
B2 724.56
B1 725.33
B7 723.09
724.96 B4
MW202
MW103 (725.96)
(716.23) G109A
G112 716.64
G109 716.39
B12 716.47
715.76 G114
G110 713.56
G113A (716.02)
G113 716.05
B13 712.34
P6 (711.98)
G115 712.58
B14 710.92
G111 715.03
B9 718.07
(721.20) MW105 MW201A MW201B B6D B6S
(719.60) (720.97)
B8 (718.32) (CLAY)
B3 (739.00) (CLAY)
P8 P9 (724.06)
B5 722.62
MW104 (722.36)
708
710
712
714
716
718
720
722
724
north
0 600 1200
SCALE IN FEET
Designed By: Drawn By: ELR Checked By: MJH
Approved By: Date: 4-01-91
Reference:
File Path: \13160\13160-BR
WARZYN
WARZYN ENGINEERING INC.
Date - By - App'd
Revisions
POTENTIOMETRIC MAP (JUNE 8, 1989)
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
Project Number
13160B15
WARZYN

FIGURE 4-9

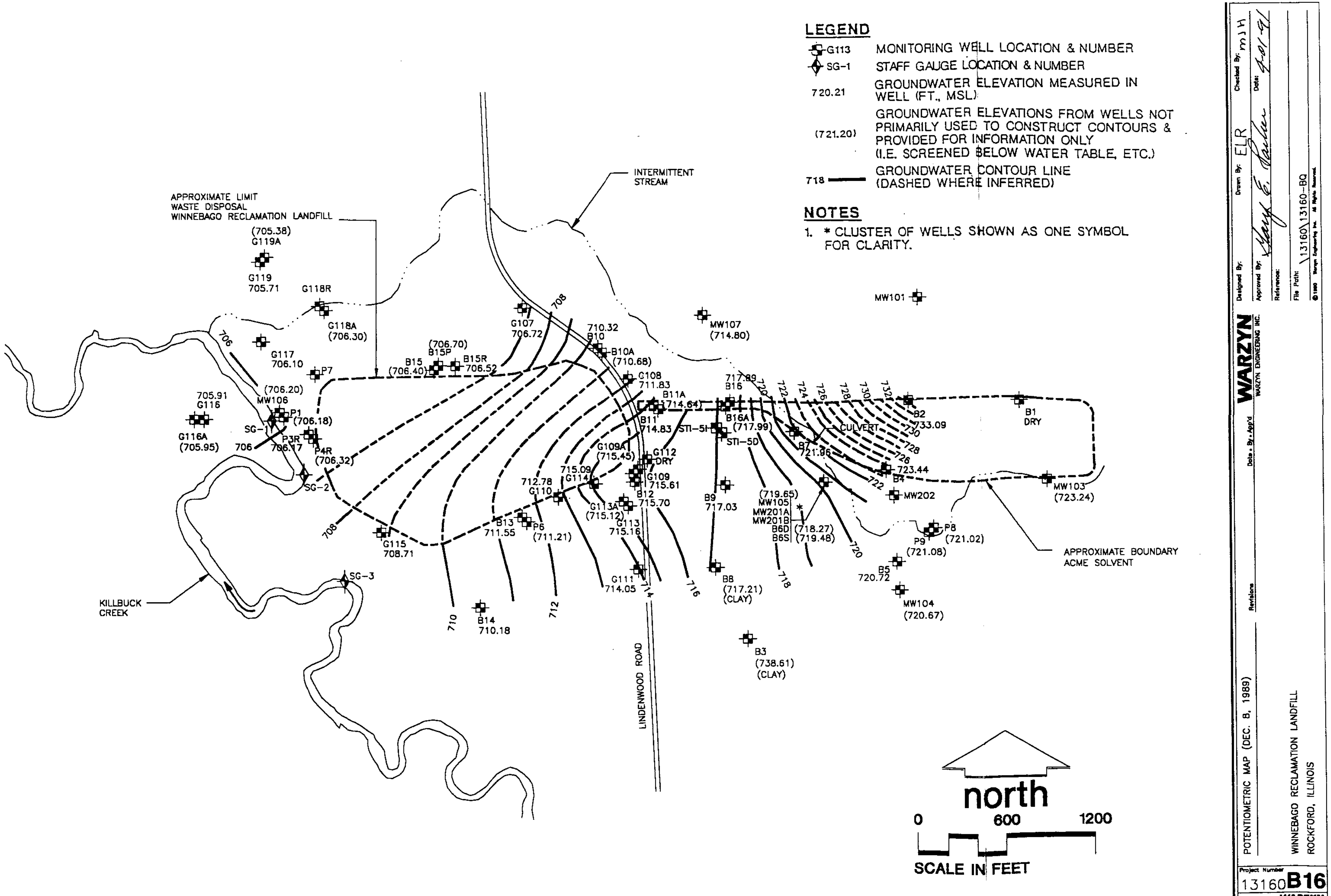

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
SG-1 STAFF GAUGE LOCATION & NUMBER
720.21 GROUNDWATER ELEVATION MEASURED IN WELL (FT., MSL)
(721.20) GROUNDWATER ELEVATIONS FROM WELLS NOT PRIMARILY USED TO CONSTRUCT CONTOURS & PROVIDED FOR INFORMATION ONLY (I.E. SCREENED BELOW WATER TABLE, ETC.)
718 GROUNDWATER CONTOUR LINE (DASHED WHERE INFERRED)
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
APPROXIMATE LIMIT WASTE DISPOSAL WINNEBAGO RECLAMATION LANDFILL
INTERMITTENT STREAM
KILLBUCK CREEK
LINDENWOOD ROAD
CULVERT
APPROXIMATE BOUNDARY ACME SOLVENT
(705.38) G119A
G119 705.71
G118R
G118A (706.30)
G117 706.10
P7
705.91 G116
G116A (705.95)
(706.20) MW106
P1 (706.18)
SG-1
P3R 706.17
P4R (706.32)
SG-2
SG-3
G107 706.72
(706.70) B15P
B15 (706.40)
B15R 706.52
710.32 B10
B10A (710.68)
G108 711.83
B11A (714.64)
B11 714.83
STI-5I
STI-5D
717.89 B16
B16A (717.99)
MW107 (714.80)
MW101
B2 733.09
B1 DRY
B7 721.86
B4 723.44
MW202
MW103 (723.24)
G109A (715.45)
G112 DRY
G109 715.61
715.09 G114
712.78 G110
B12 715.70
G113A (715.12)
G113 715.16
B13 711.55
P6 (711.21)
G115 708.71
B9 717.03
(719.65) MW105
MW201A
MW201B
B6D (718.27)
B6S (719.48)
P8 (721.02)
P9 (721.08)
B5 720.72
MW104 (720.67)
G111 714.05
B8 (717.21) (CLAY)
B14 710.18
B3 (738.61) (CLAY)
706 708 710 712 714 716 718 720 722 724 726 728 730 732
north
0 600 1200
SCALE IN FEET
WARZYN
WARZYN ENGINEERING INC.
Designed By:
Drawn By: ELR
Checked By: mJH
Approved By:
Date:
Reference:
File Path: 13160\13160-BQ
Revisions
POTENTIOMETRIC MAP (DEC. 8, 1989)
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
Project Number
13160B16
WARZYN

FIGURE 4-10

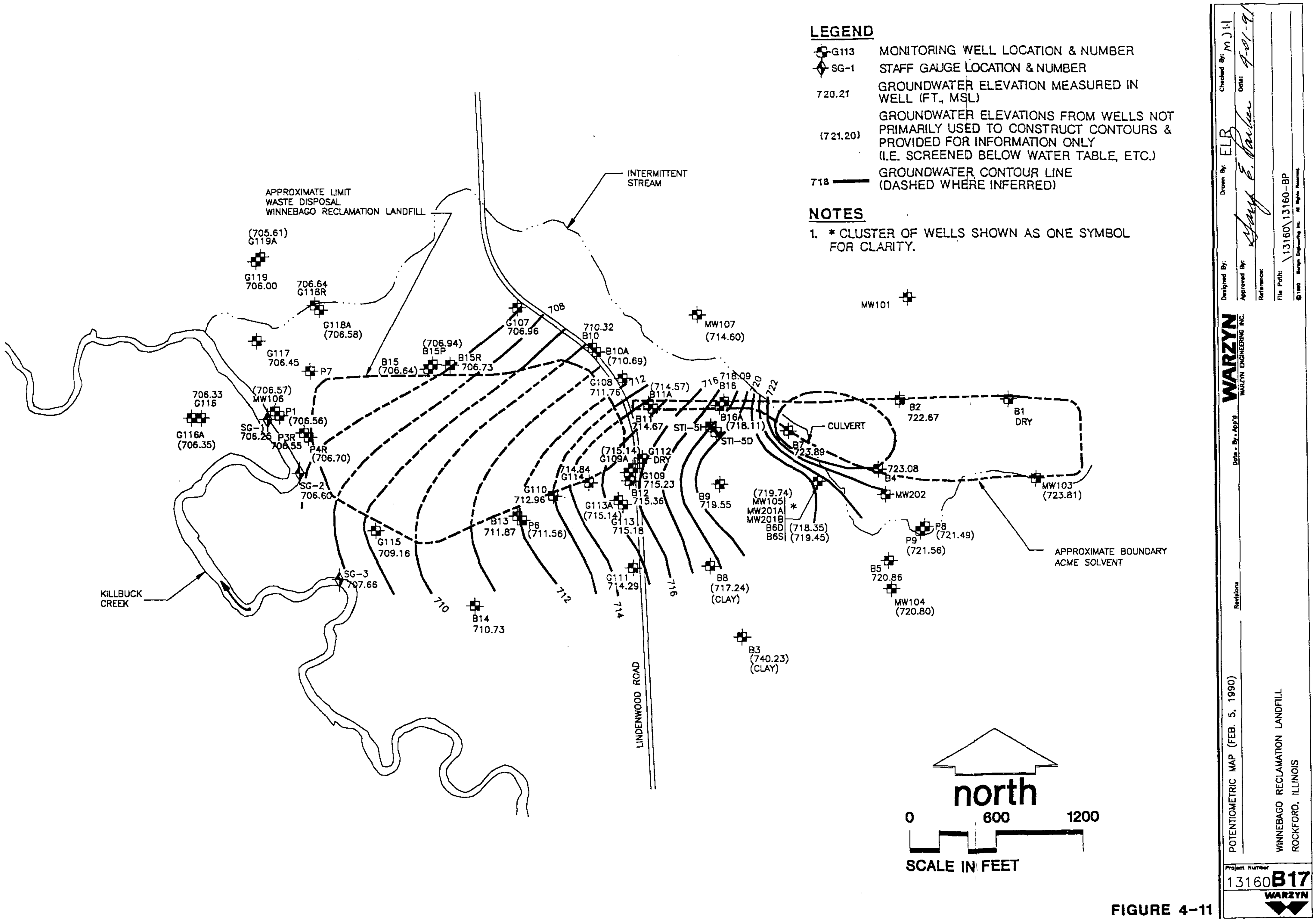
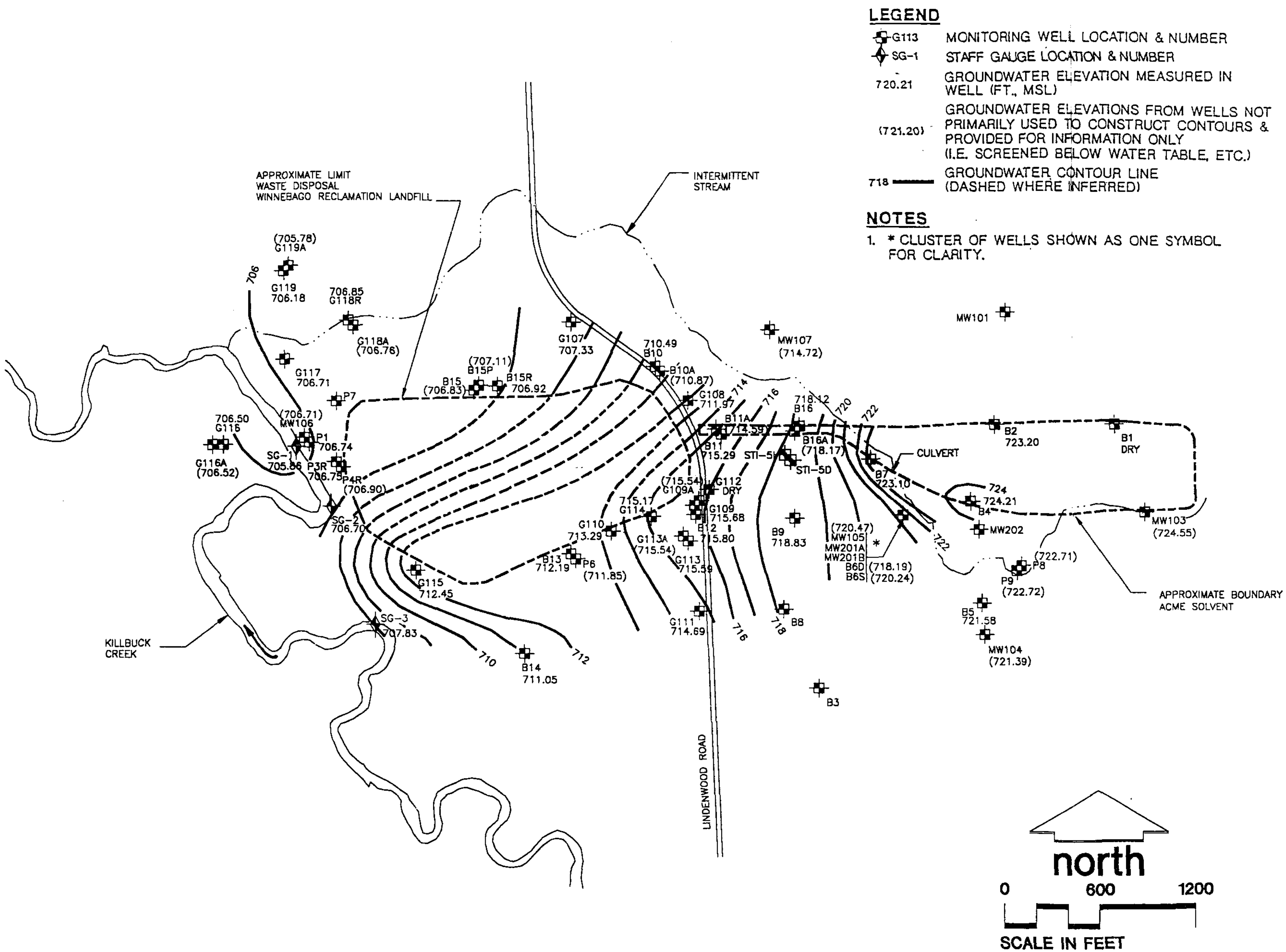

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
SG-1 STAFF GAUGE LOCATION & NUMBER
720.21 GROUNDWATER ELEVATION MEASURED IN WELL (FT., MSL)
(721.20) GROUNDWATER ELEVATIONS FROM WELLS NOT PRIMARILY USED TO CONSTRUCT CONTOURS & PROVIDED FOR INFORMATION ONLY (I.E. SCREENED BELOW WATER TABLE, ETC.)
718 GROUNDWATER CONTOUR LINE (DASHED WHERE INFERRED)
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
INTERMITTENT STREAM
APPROXIMATE LIMIT WASTE DISPOSAL WINNEBAGO RECLAMATION LANDFILL
(705.61) G119A
G119 706.00
706.64 G118R
G118A (706.58)
G117 706.45
P7
706.33 G116
G116A (706.35)
(706.57) MW106
P1
(706.56)
SG-1 706.25
P3R 706.55
P4R (706.70)
SG-2 706.60
SG-3 707.66
KILLBUCK CREEK
B15 (706.64)
(706.94) B15P
B15R 706.73
G107 706.96
708
710.32 B10
B10A (710.69)
G108 711.76
712
(714.57) B11A
B11 714.67
716
718.09 B16
720
722
B16A (718.11)
STI-5H
STI-5D
MW107 (714.60)
MW101
CULVERT
B2 722.67
B1 DRY
B7 723.89
723.08 B4
MW202
MW103 (723.81)
(715.14) G109A
G112 DRY
714.84 G114
G109 715.23
B12 715.36
G110 712.96
G113A (715.14)
G113 715.18
B9 719.55
(719.74) MW105 MW201A MW201B B6D B6S
(718.35) (719.45)
P8 (721.49)
P9 (721.56)
B5 720.86
MW104 (720.80)
APPROXIMATE BOUNDARY ACME SOLVENT
B13 711.87
P6 (711.56)
G115 709.16
710
B14 710.73
712
714
G111 714.29
716
B8 (717.24) (CLAY)
B3 (740.23) (CLAY)
LINDENWOOD ROAD
north
0 600 1200
SCALE IN FEET
POTENTIOMETRIC MAP (FEB. 5, 1990)
WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS
Designed By: Drawn By: ELB Checked By: MJH
Approved By: Date: 4-01-91
File Path: \13160\13160-BP
WARZYN
WARZYN ENGINEERING INC.
Revisions
Project Number 13160 B17

FIGURE 4-11



POTENTIOMETRIC MAP (MARCH 5, 1990)

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

WARZYN
WARZYN ENGINEERING INC.

Drawn By: ELR
Checked By: MJH
Approved By: Mary E. Parker
Date: 4-01-91
Designed By:
Reference:
File Path: \13160\13160-B0

Revisions
Date - By - App'd

Project Number
13160B18
WARZYN

FIGURE 4-12

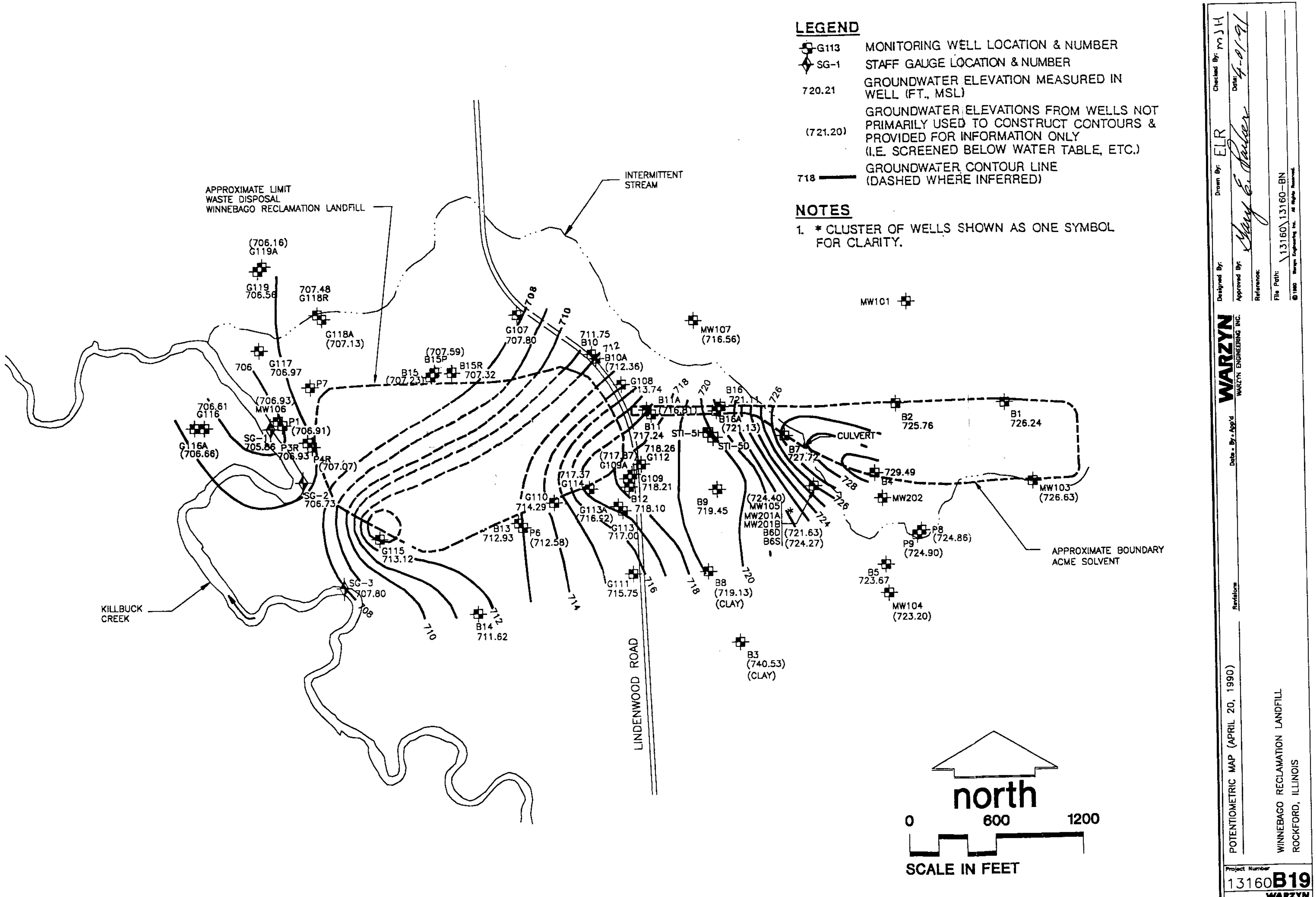
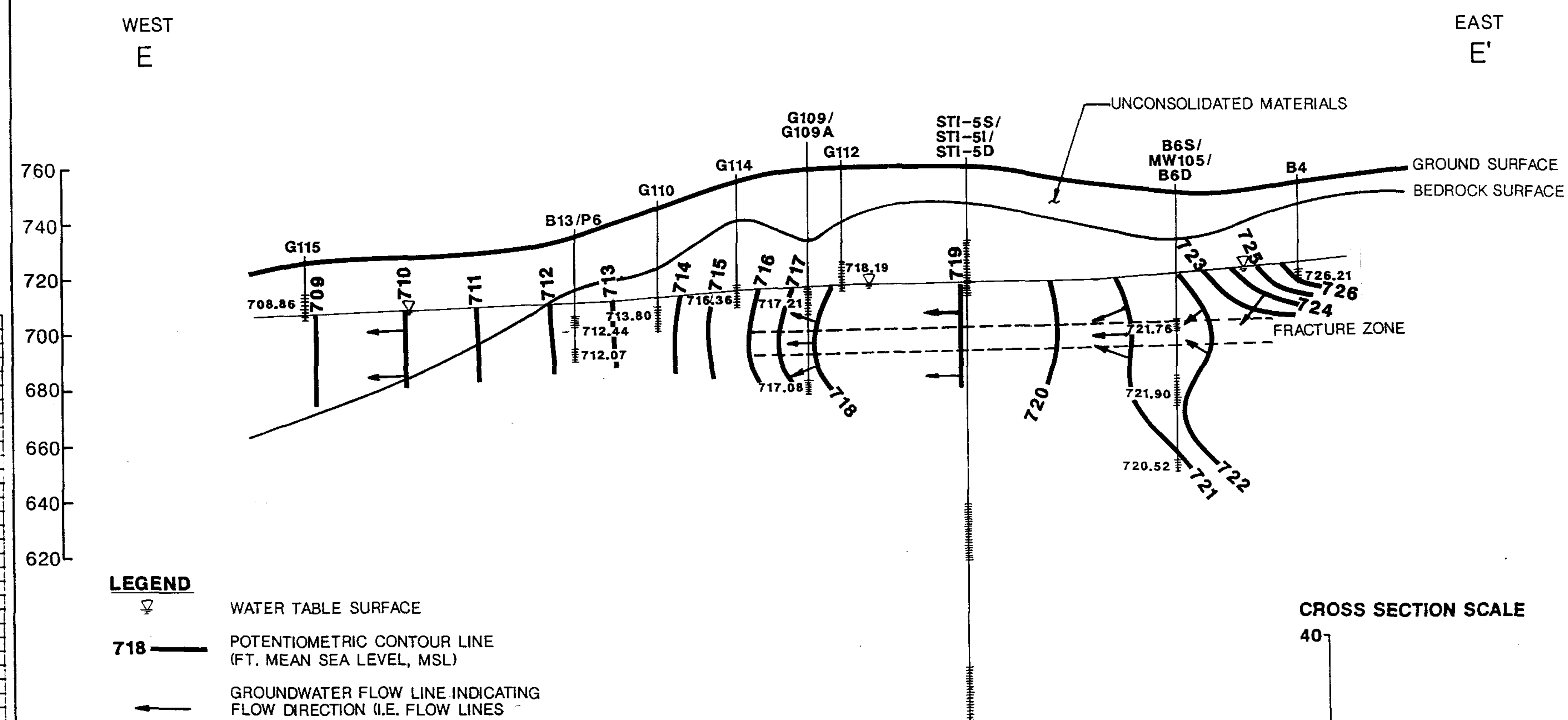

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
SG-1 STAFF GAUGE LOCATION & NUMBER
720.21 GROUNDWATER ELEVATION MEASURED IN WELL (FT., MSL)
(721.20) GROUNDWATER ELEVATIONS FROM WELLS NOT PRIMARILY USED TO CONSTRUCT CONTOURS & PROVIDED FOR INFORMATION ONLY (I.E. SCREENED BELOW WATER TABLE, ETC.)
718 GROUNDWATER CONTOUR LINE (DASHED WHERE INFERRED)
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
APPROXIMATE LIMIT WASTE DISPOSAL WINNEBAGO RECLAMATION LANDFILL
INTERMITTENT STREAM
KILLBUCK CREEK
LINDENWOOD ROAD
CULVERT
APPROXIMATE BOUNDARY ACME SOLVENT
north
0 600 1200
SCALE IN FEET
POTENTIOMETRIC MAP (APRIL 20, 1990)
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
WARZYN
WARZYN ENGINEERING INC.
Project Number 13160B19
File Path: \13160\13160-BN

FIGURE 4-13



LEGEND

WATER TABLE SURFACE

718 POTENTIOMETRIC CONTOUR LINE (FT. MEAN SEA LEVEL, MSL)

GROUNDWATER FLOW LINE INDICATING FLOW DIRECTION (I.E. FLOW LINES ARE PERPENDICULAR LINES)

ESTIMATED BOUNDARY OF FRACTURE ZONE

G115 WELL NUMBER

718.19 WELL SCREEN/INTAKE INTERVAL AND WATER LEVEL ELEVATION (FT., MSL)

NOTES

1. WATER LEVEL ELEVATIONS FROM OCTOBER 25, 1988.
2. SEE FIGURE 4-26 FOR CROSS-SECTION LOCATION.
3. SEE TEXT FOR DISCUSSION OF FRACTURE ZONE.

CROSS SECTION SCALE

40

0

0 400 800

SCALE IN FEET

VERTICAL EXAGGERATION: 10 TIMES

POTENTIOMETRIC CROSS-SECTION E-E'

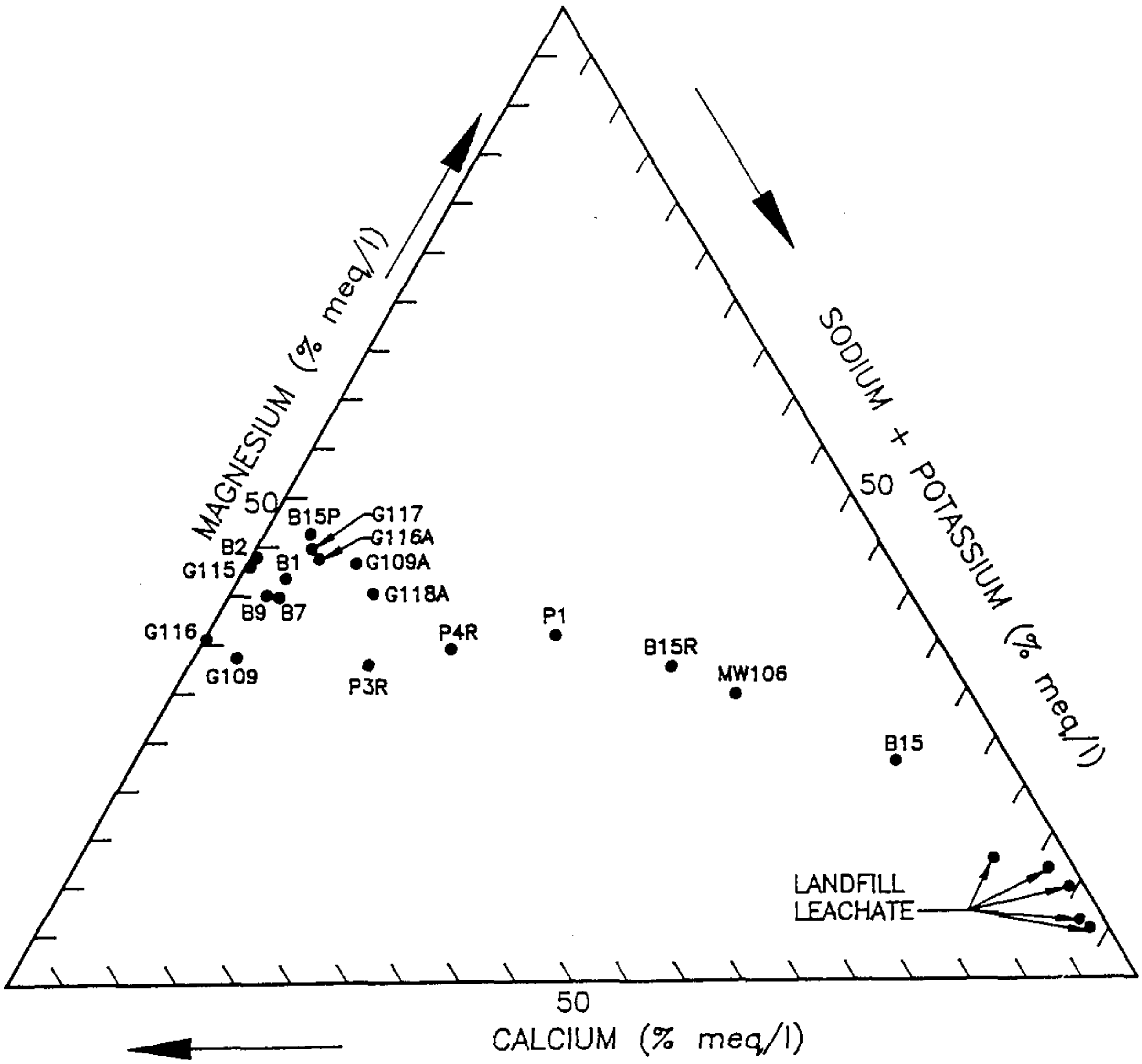
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

WARZYN
WARZYN ENGINEERING INC.

Drawn By: D.L.L. Checked By: mJ.l
Approved By: Gary E. Parker Date: 4-01-91

Project Number 13160 B20

FIGURE 4-19



NOTES

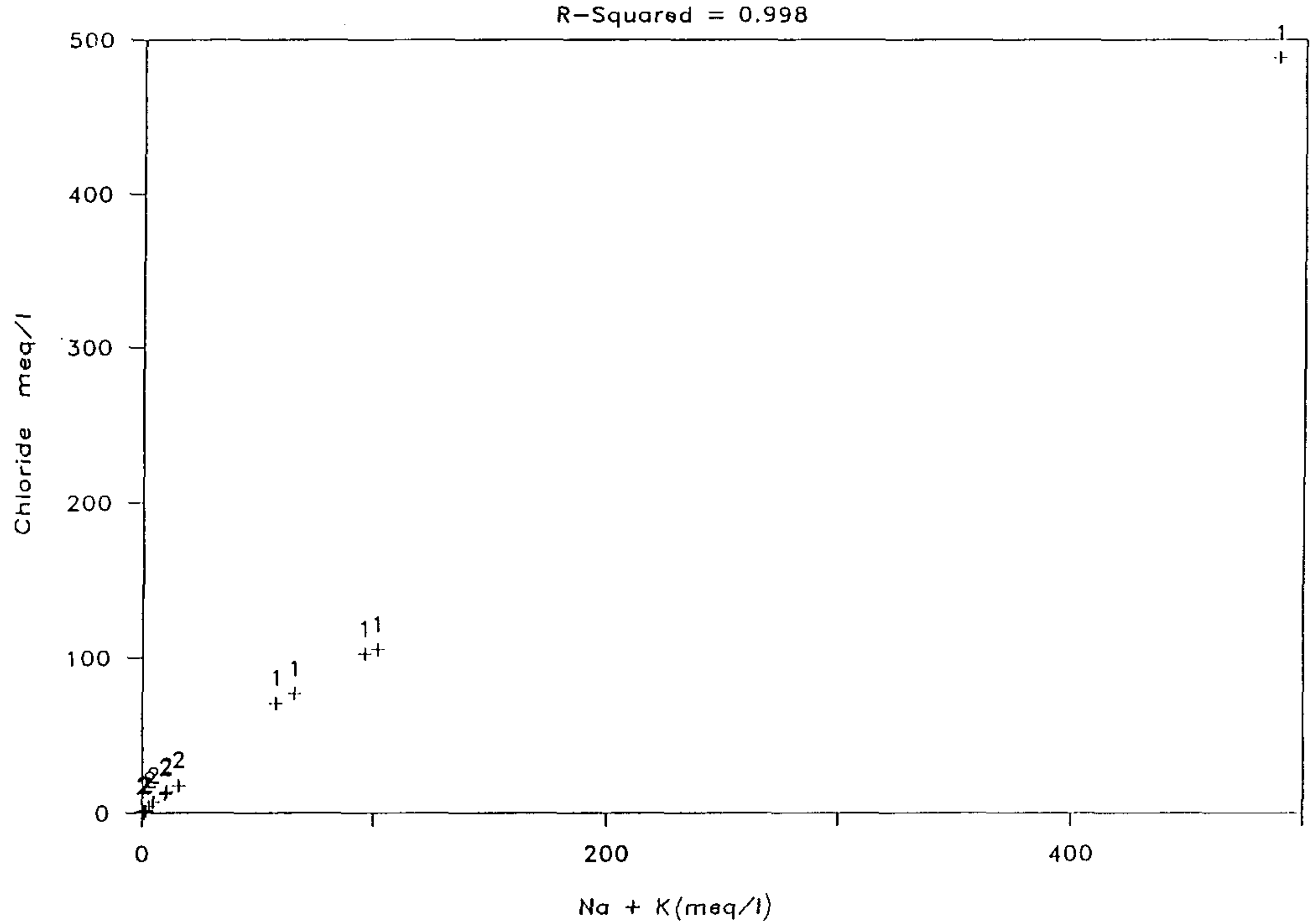
1. ALL DATA FROM ROUND 1 SAMPLING EXCEPT FOR B1, B2, B7, AND B9 WHICH ARE FROM AUGUST 3-7, 1988 ACME SOLVENT RI DATA PROVIDED BY THE U.S. EPA.

FIGURE 4-20

DWN FLR	APP'D	DATE 11-22-88	13160	A12	WARZYN ENGINEERING INC	MAJOR CATION TRILINEAR PLOT WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS

NOTES

1. ROUND ONE LEACHATE SAMPLES A1, E7, F1, K2, MHW IDENTIFIED AS 1, ROUND ONE MONITORING WELL SAMPLES B15, B15P, B15R, G109, G109A, G115, G116, G116A, G117, G118A, MW106, P1, P3R & P4R IDENTIFIED AS 2.

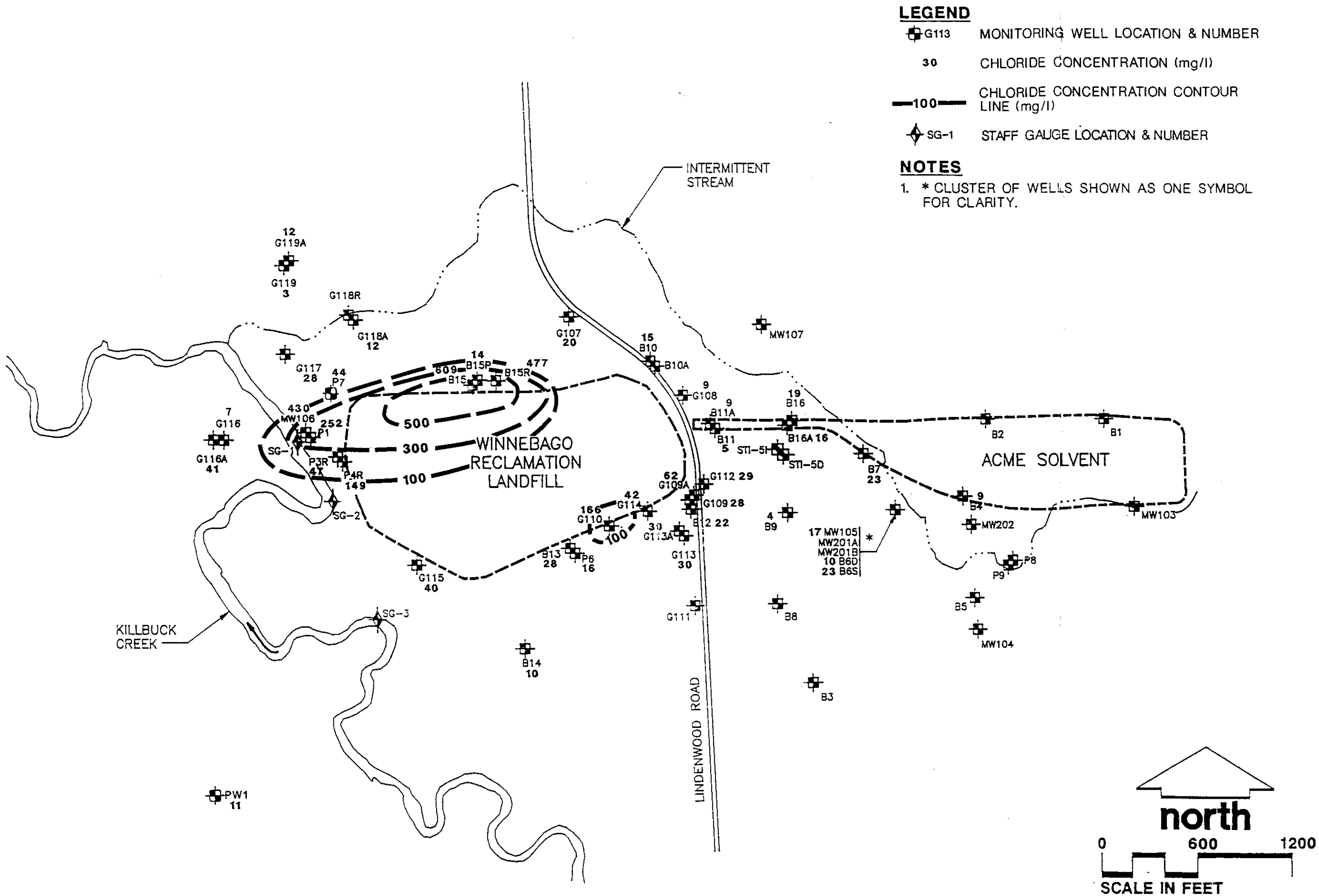


DWN. ELR APP'D DATE 1-22-89 13160 A13

WARZYN ENGINEERING INC

CHLORIDE vs SODIUM & POTASSIUM
WINNEBAGO RECLAMATION LANDFILL
ROCFORD, ILLINOIS

FIGURE 4-21



WARZYN
WARZYN ENGINEERING INC.

Drawn By: ELR, DLL. Checked By: MJH
Approved By: [signature] Date: 9/01/91

ROUND 1 CHLORIDES

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

Project Number 13160 B21

FIGURE 4-22

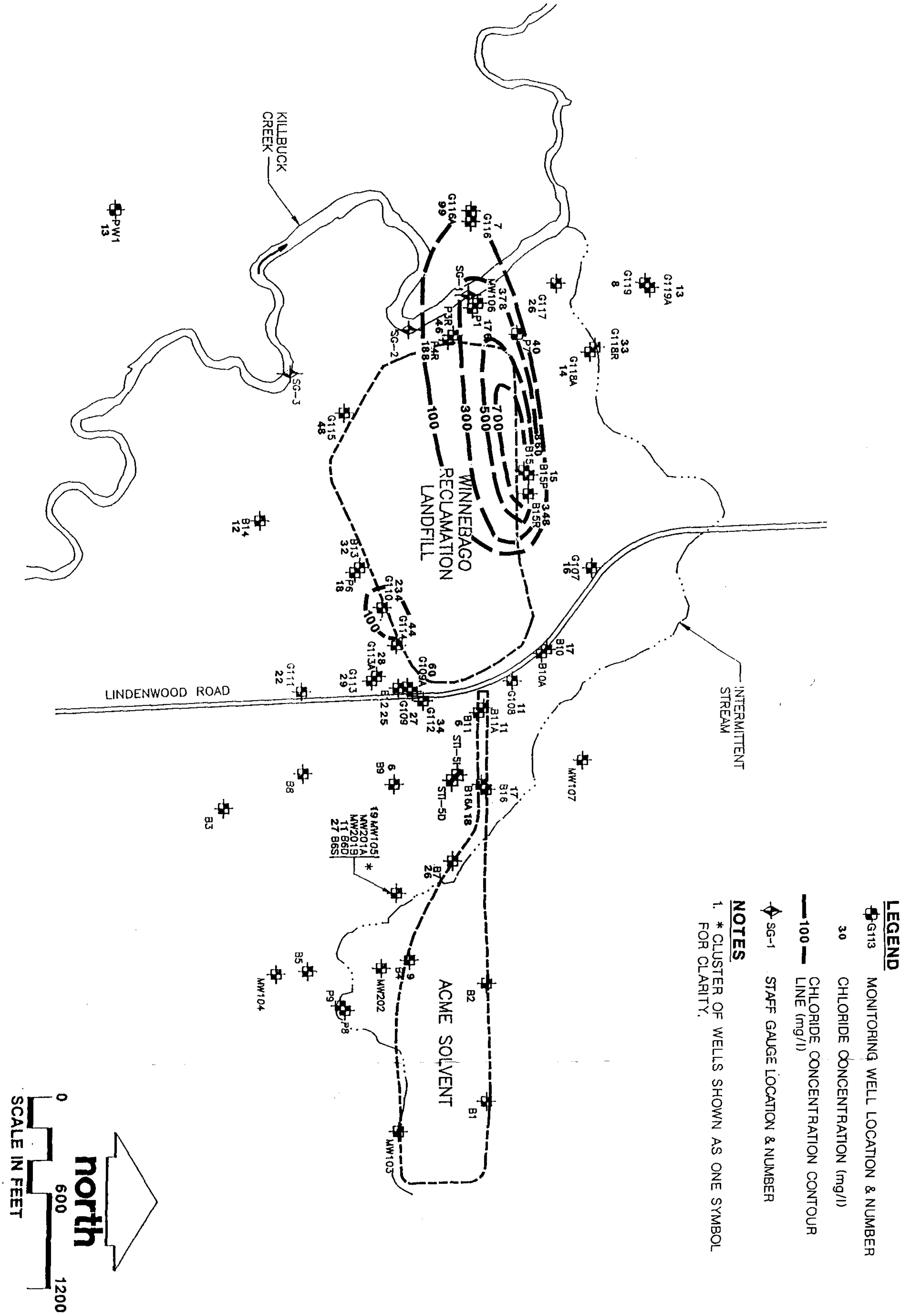
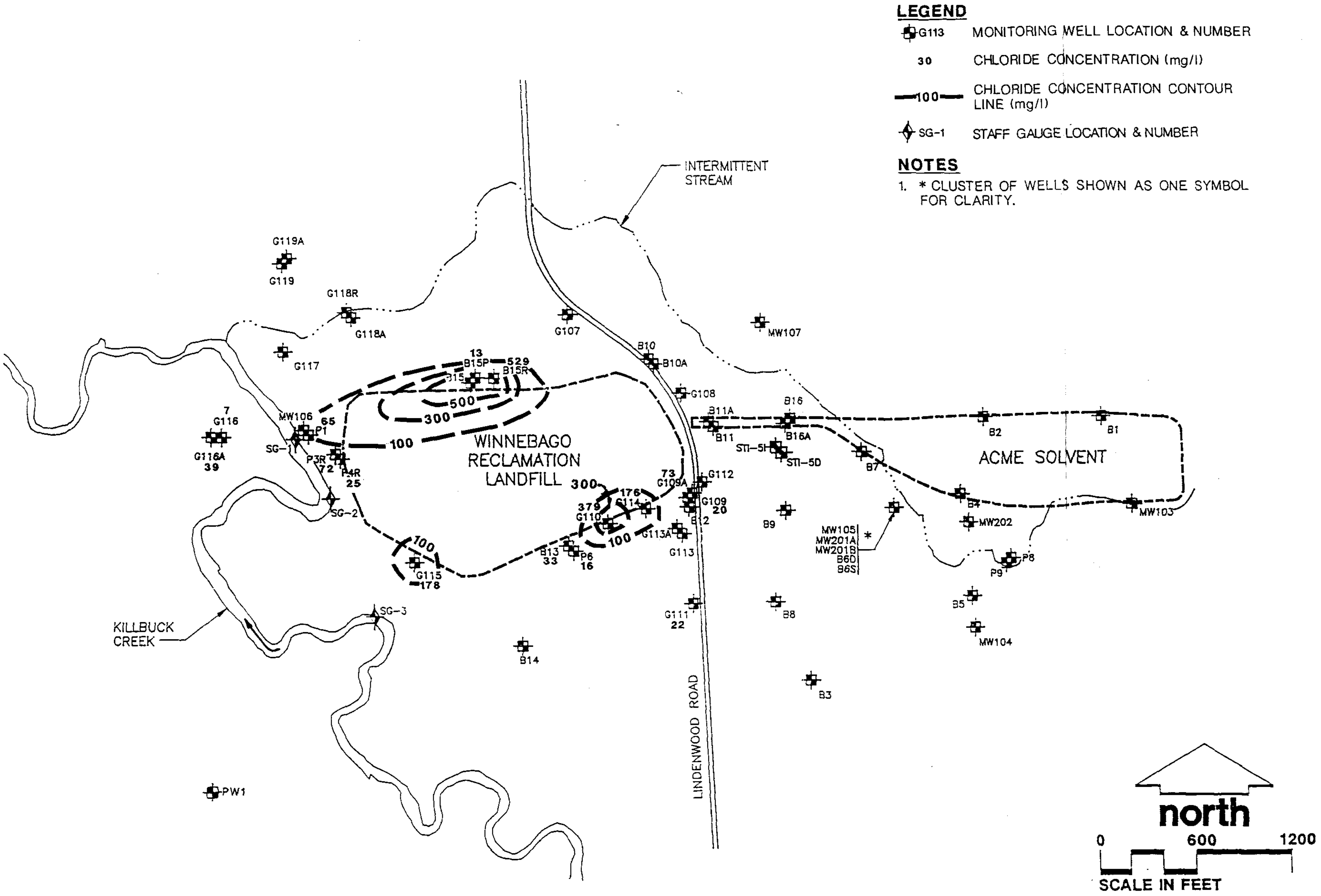


FIGURE 4-23

ROUND 2 CHLORIDES WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS	Revisions / Date / By / App'd	WARZYN WARZYN ENGINEERING INC.
Designed By:	Drawn By: ELR, DLL.	Checked By: MJH
Approved By: Gary E. Parker		Date: 4/01/91
Reference:		
File Path:		

Project Number 13160 B22



ROUND 3 CHLORIDES

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

WARZYN

Drawn By: ELR, DLL. Checked By: MJH

Project Number 13160 B23

FIGURE 4-24

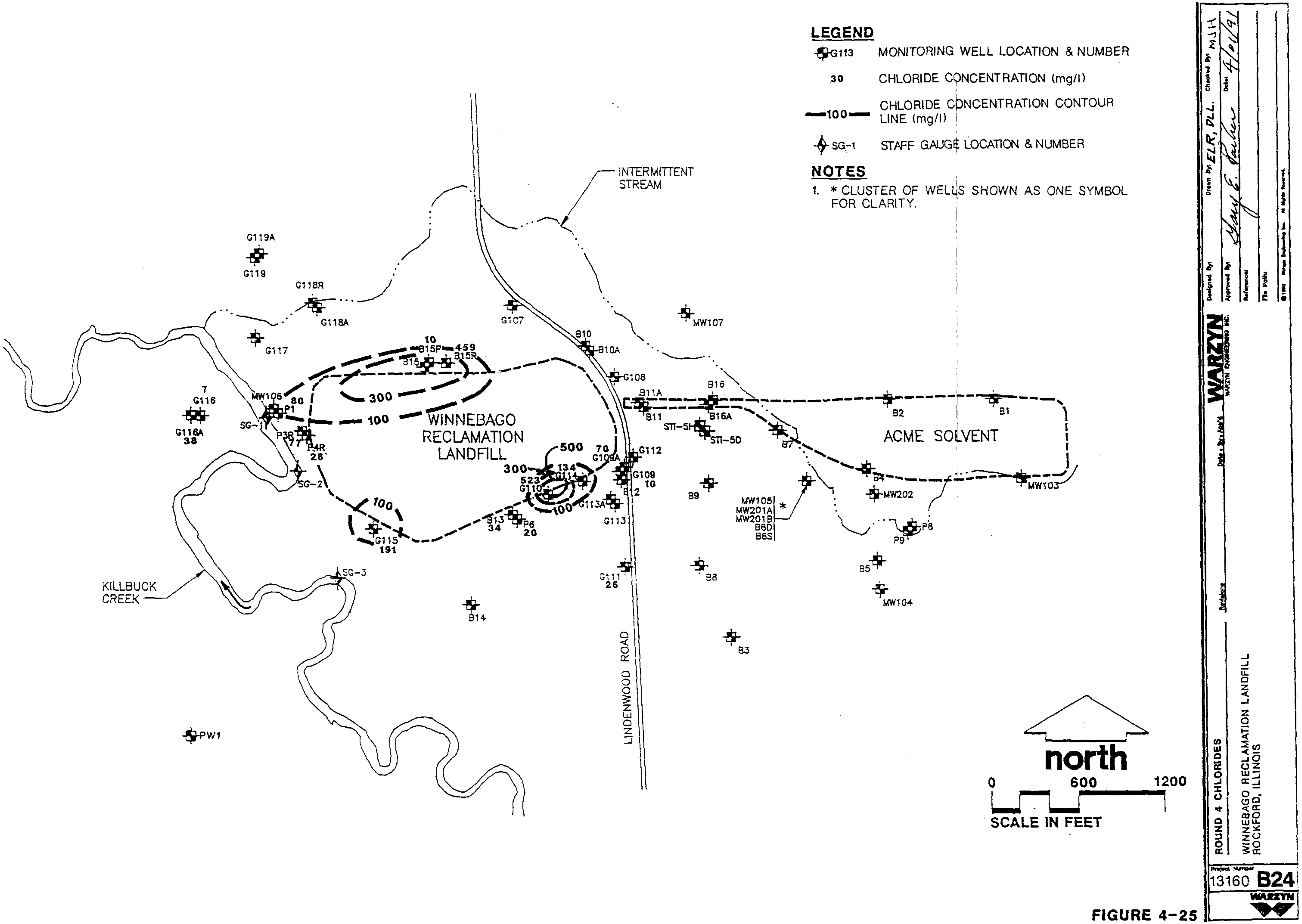

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
30 CHLORIDE CONCENTRATION (mg/l)
100 CHLORIDE CONCENTRATION CONTOUR LINE (mg/l)
SG-1 STAFF GAUGE LOCATION & NUMBER
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
INTERMITTENT STREAM
WINNEBAGO RECLAMATION LANDFILL
ACME SOLVENT
KILLBUCK CREEK
LINDENWOOD ROAD
G119A
G119
G118R
G118A
G117
G107
MW107
B10
B10A
G108
G116
7
G116A
38
MW106
80
P1
SG-1
P3R
77
P4R
28
SG-2
10
B15F
459
B15R
B15
300
100
500
300
134
523
G114
G110
100
70
G109A
G112
G109
10
B12
G113A
G113
B13
34
P6
20
G115
191
100
SG-3
B14
G111
26
B11A
B11
B16
B16A
STI-5H
STI-5D
B7
B2
B1
B4
MW103
B9
MW105
MW201A
MW201B
B6D
B6S
*
MW202
P8
P9
B5
MW104
B8
B3
PW1
north
0
600
1200
SCALE IN FEET
WARZYN
WARZYN ENGINEERING INC.
Drawn By: ELR, DLL.
Checked By: MJH
ROUND 4 CHLORIDES
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
13160
B24

FIGURE 4-25

LEGEND

- G113 — MONITORING WELL LOCATION & NUMBER
- SG-1 — STAFF GAUGE LOCATION & NUMBER

NOTES

1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.

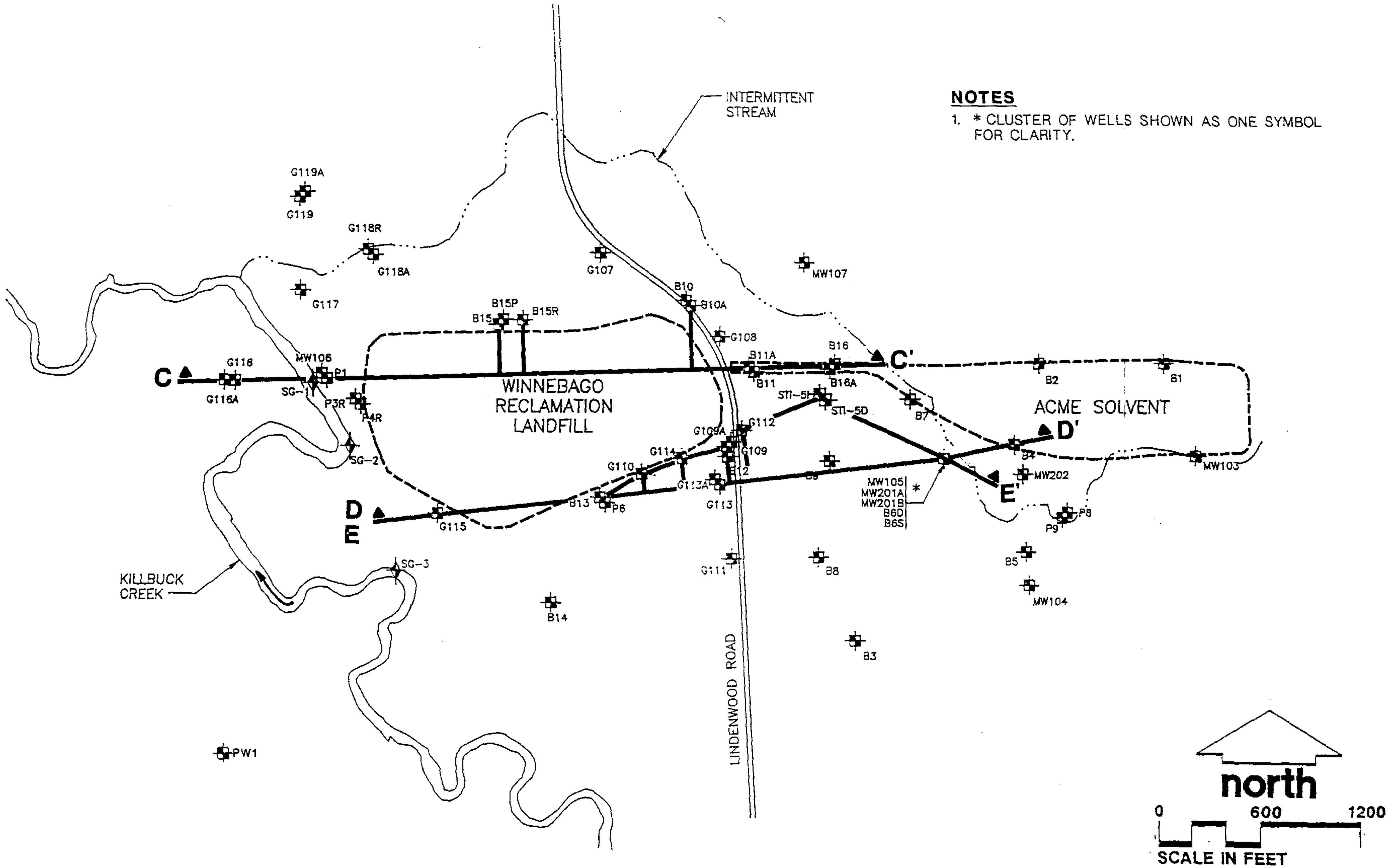
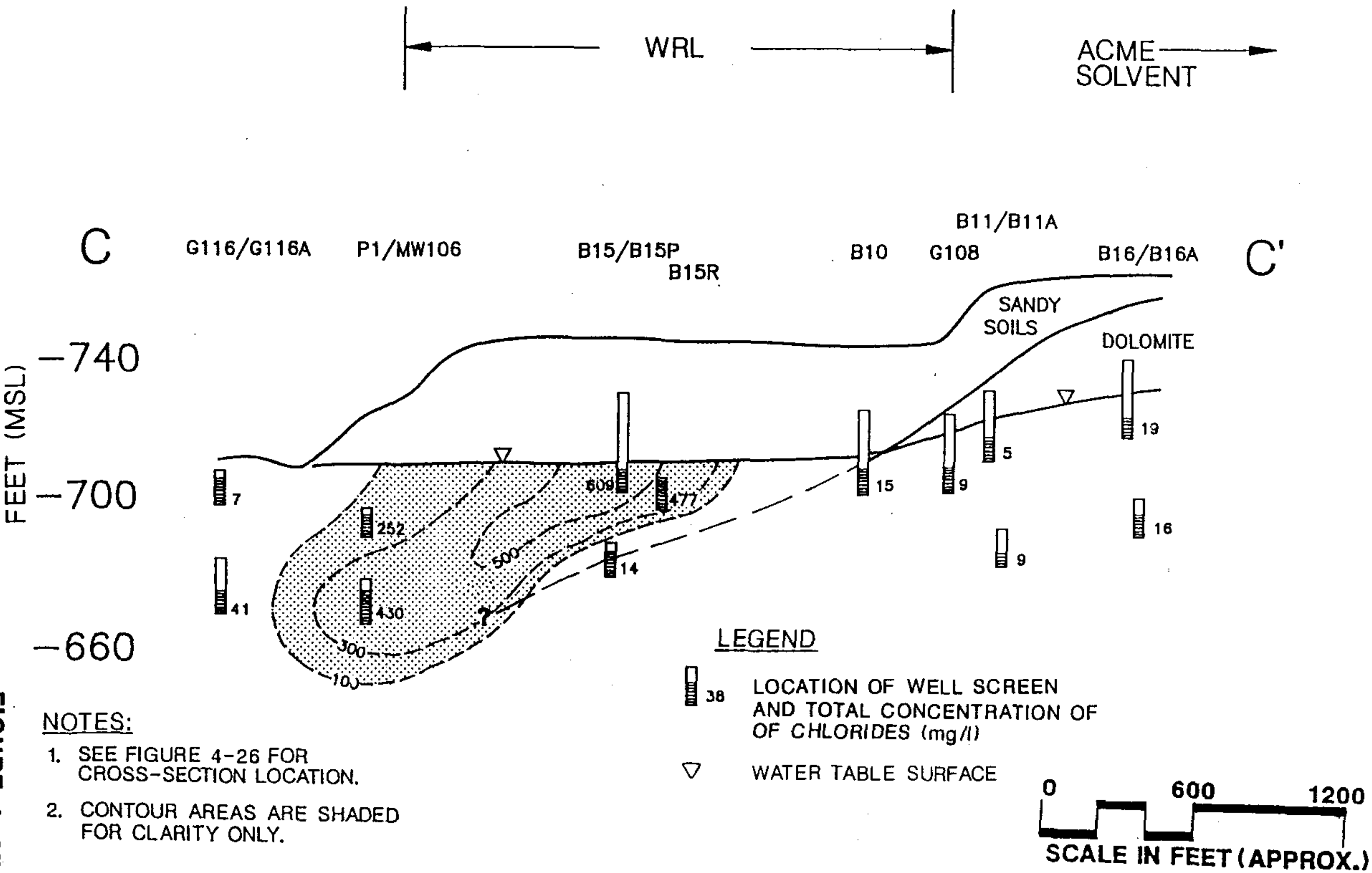


FIGURE 4-26

CROSS-SECTION C-C', D-D' & E-E' LOCATION MAP	Revisions / Date / By / App'd
WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS	WARZYN WARZYN ENGINEERING INC.
Project Number 13160 B25	Designed By: / Drawn By: ELR, DLL. / Checked By: MJH
	Approved By: Gary E. Parker / Date: 4/01/91
	Reference:
	File Path:



NOTES:

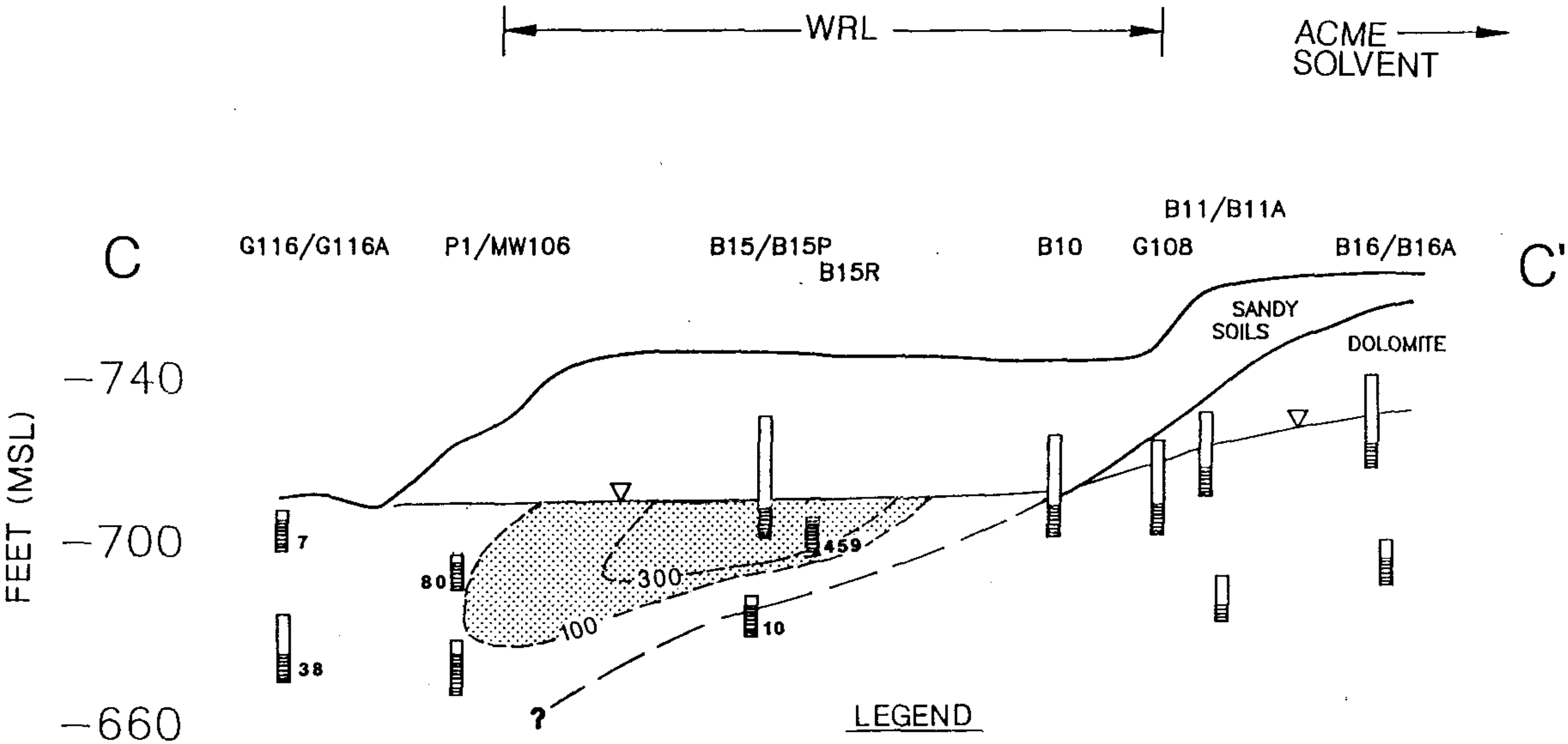
1. SEE FIGURE 4-26 FOR CROSS-SECTION LOCATION.
2. CONTOUR AREAS ARE SHADED FOR CLARITY ONLY.

WARZYN

FIGURE 4-27

ROUND 1 CHLORIDES (mg/l)

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS



NOTES:

1. SEE FIGURE 4-26 FOR CROSS-SECTION LOCATION.
2. CONTOUR AREAS ARE SHADED FOR CLARITY ONLY.

LEGEND

38 LOCATION OF WELL SCREEN AND TOTAL CONCENTRATION OF CHLORIDES (mg/l)

▽ WATER TABLE SURFACE

0 600 1200
SCALE IN FEET (APPROX.)

DWN. ELR | APP'D. 4EP | DATE 4-01-91 | 13160 | A16

WARZYN ENGINEERING INC

FIGURE 4-29

ROUND 4 CHLORIDES (mg/l)

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

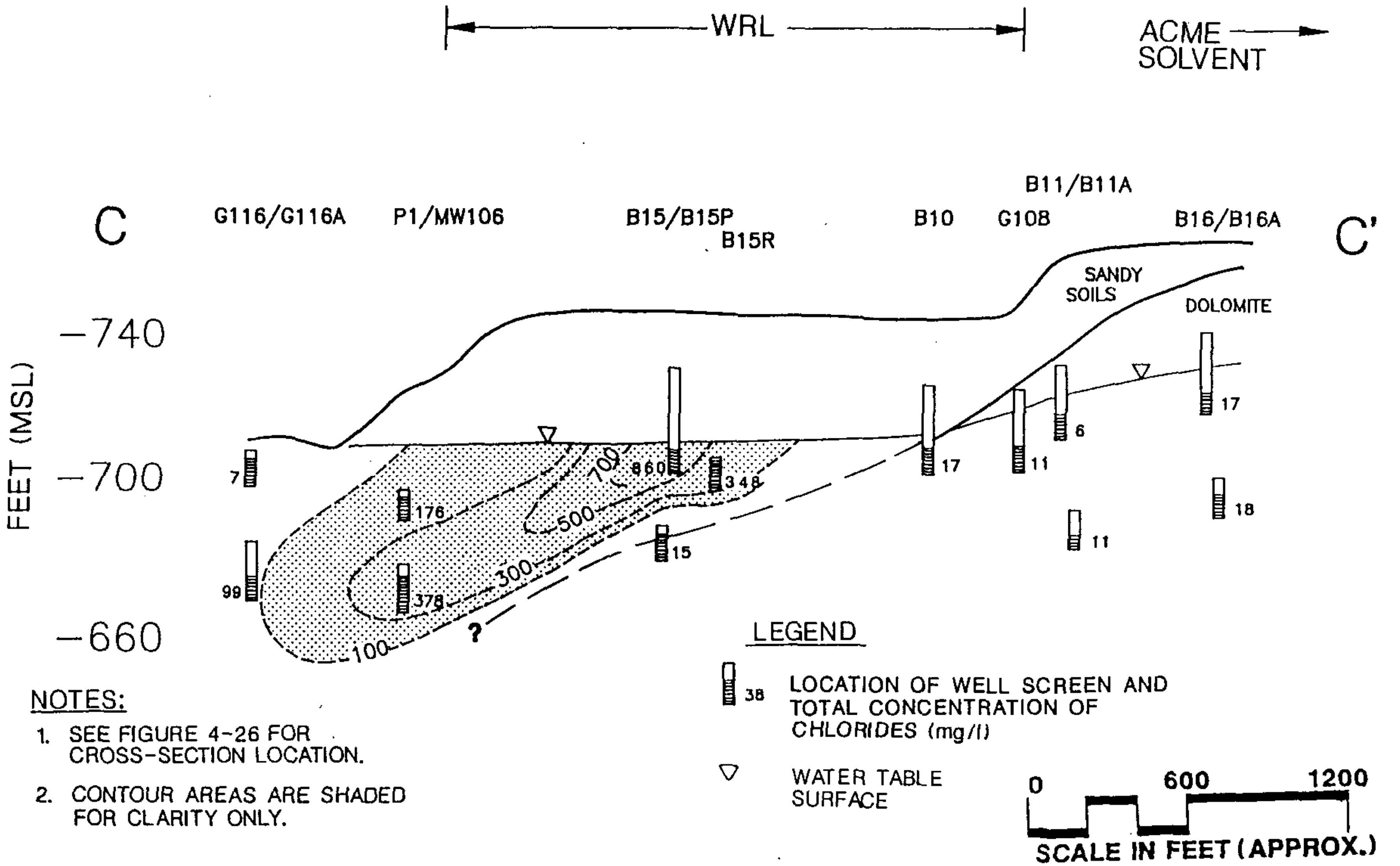


FIGURE 4-28

ROUND 2 CHLORIDES (mg/l)

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

WARZYN ENGINEERING INC

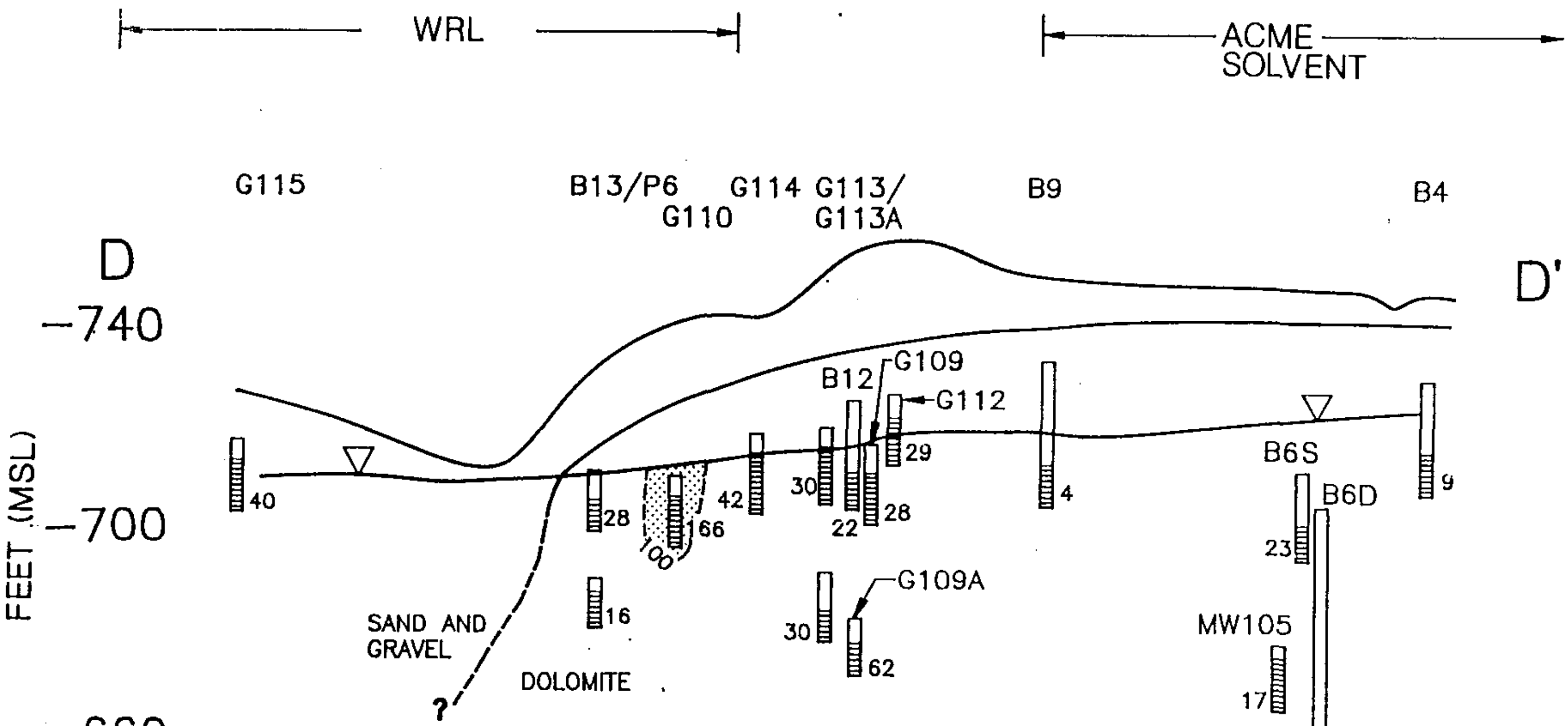
A15

13160

DWN. ELR

APP'D

DATE 11-23-



LEGEND

38 LOCATION OF WELL SCREEN AND TOTAL CONCENTRATION OF CHLORIDES (mg/l)

▽ WATER TABLE SURFACE

0 600 1200
SCALE IN FEET (APPROX.)

NOTES:

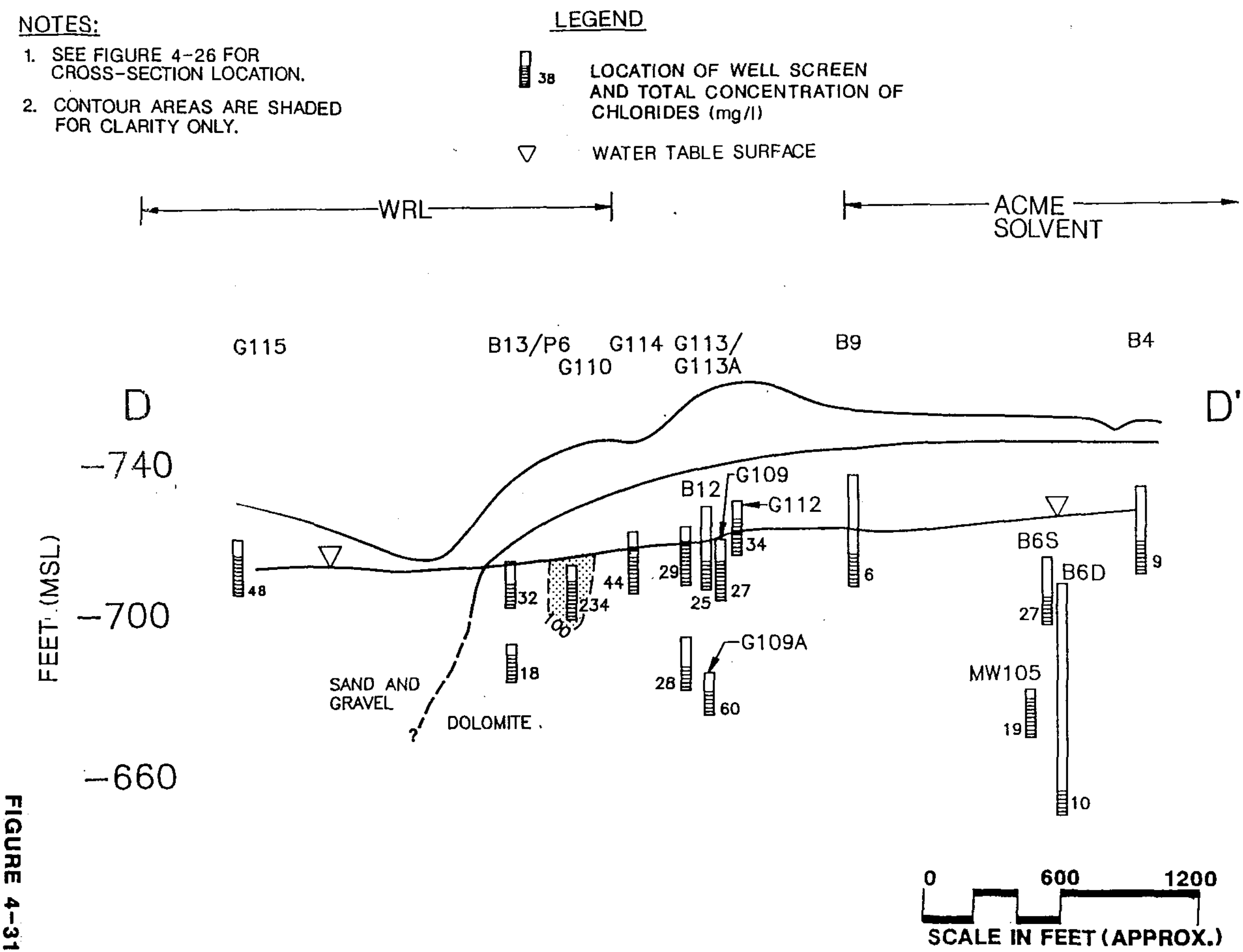
1. SEE FIGURE 4-26 FOR CROSS-SECTION LOCATION.
2. CONTOUR AREAS ARE SHADED FOR CLARITY ONLY.

WARZYN ENGINEERING INC

ROUND 1 CHLORIDES (mg/l)

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

FIGURE 4-30



DWN. ELR
APP'D
DATE 1-22-85
13160
A18
WARZYN ENGINEERING INC

ROUND 2 CHLORIDES (mg/l)
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

FIGURE 4-31

NOTES:

1. SEE FIGURE 4-26 FOR CROSS-SECTION LOCATION.
2. CONTOUR AREAS ARE SHADED FOR CLARITY ONLY.

LEGEND

38 LOCATION OF WELL SCREEN AND TOTAL CONCENTRATION OF CHLORIDES (mg/l)

▽ WATER TABLE SURFACE

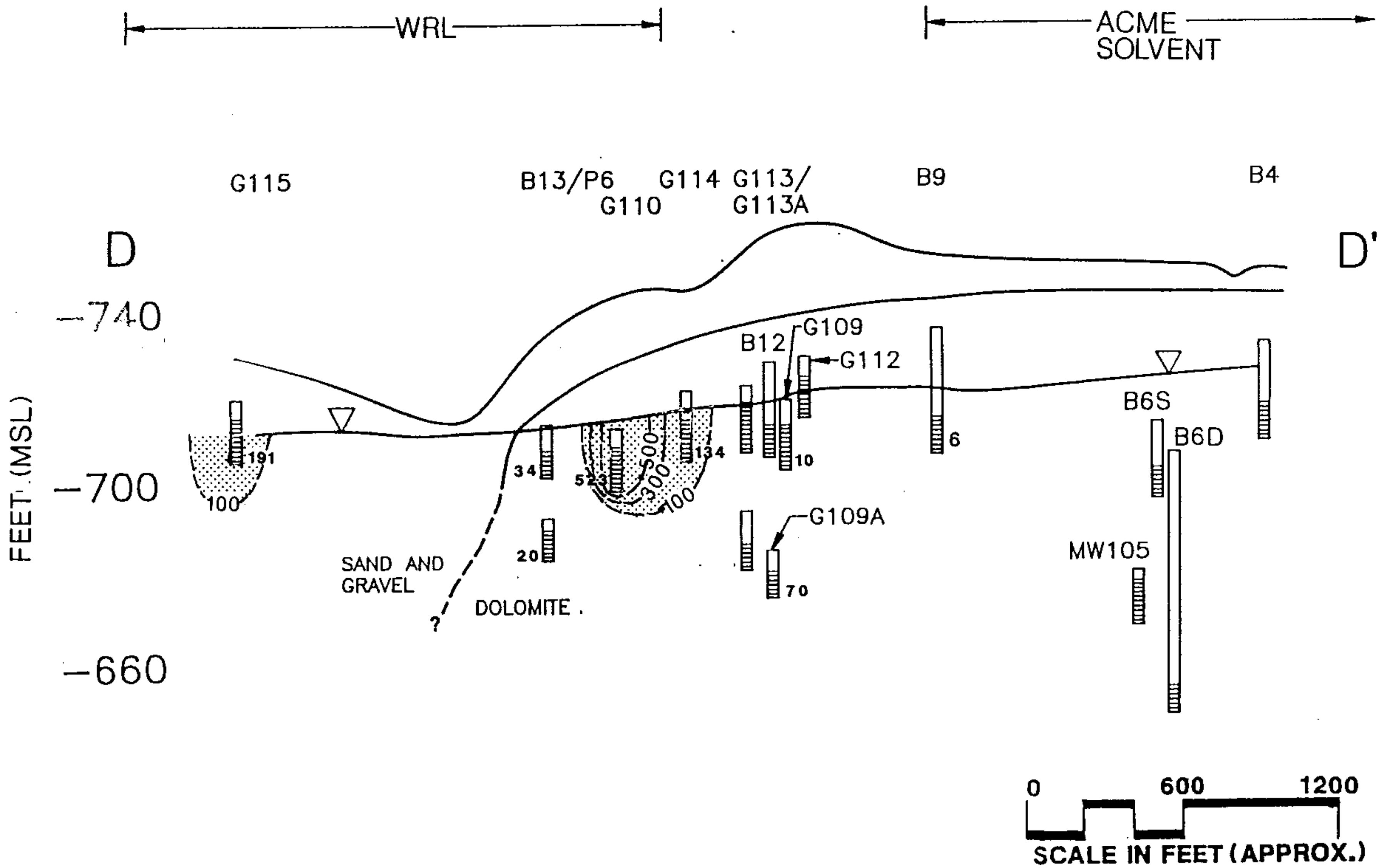
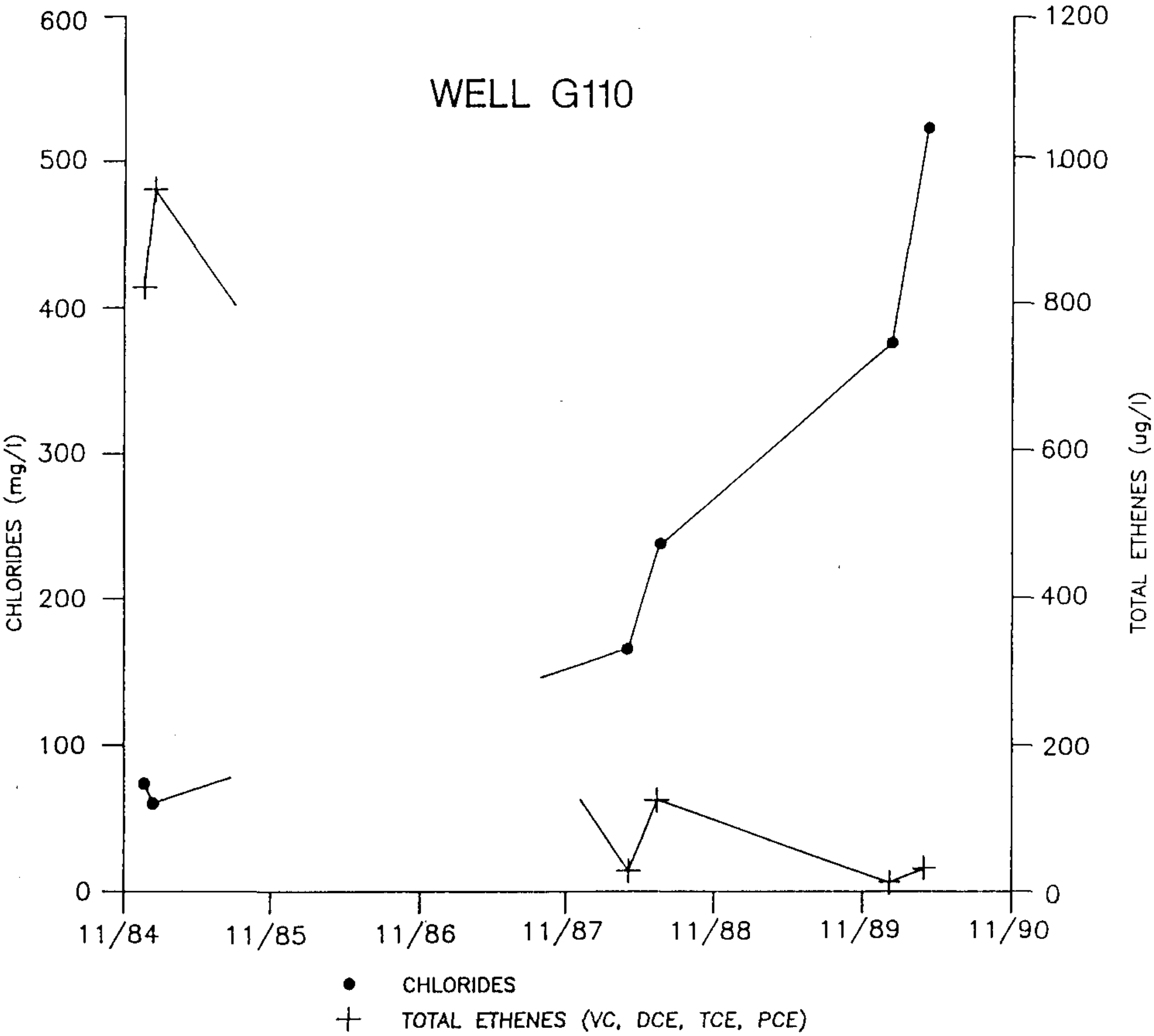


FIGURE 4-32

WARZYN

ROUND 4 CHLORIDES (mg/l)

WINNEBAGO RECLAMATION LANDFILL



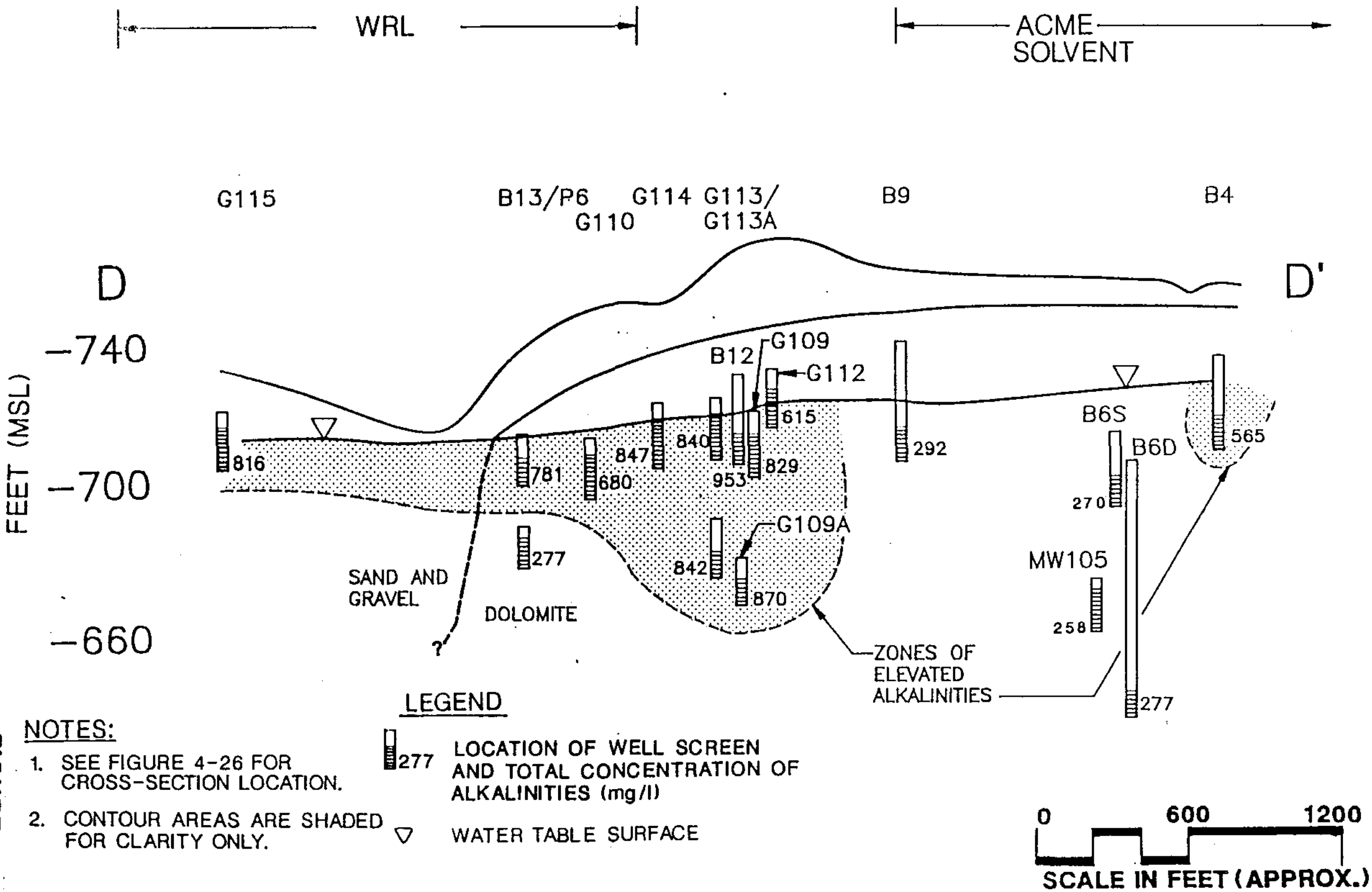
NOTES

1. DECEMBER 1984 AND JANUARY 1985 SAMPLING REPORTED IN WARZYN INC. 1985.
2. APRIL 1988 THROUGH APRIL 1990 SAMPLING REPORTED IN WARZYN INC. 1990.

QUALITY CONTROL — Drafting Standards, Lead Professional, Section, PM, Division, Other

FIGURE 4-33

WARZYN	CHLORIDES & TOTAL ETHENES	Drawn DLL	Checked MJH	App'd. HEP
	WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS			Date 4-01-91
				13160 A20



WARZYN

FIGURE 4-37

ROUND 2 ALKALINITY (mg/l)

WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS

NOTES:

1. SEE FIGURE 4-26 FOR CROSS-SECTION LOCATION.
2. CONTOUR AREAS ARE SHADED FOR CLARITY ONLY.

LEGEND

7.40 LOCATION OF WELL SCREEN & pH READING (STANDARD UNITS)

▽ WATER TABLE SURFACE

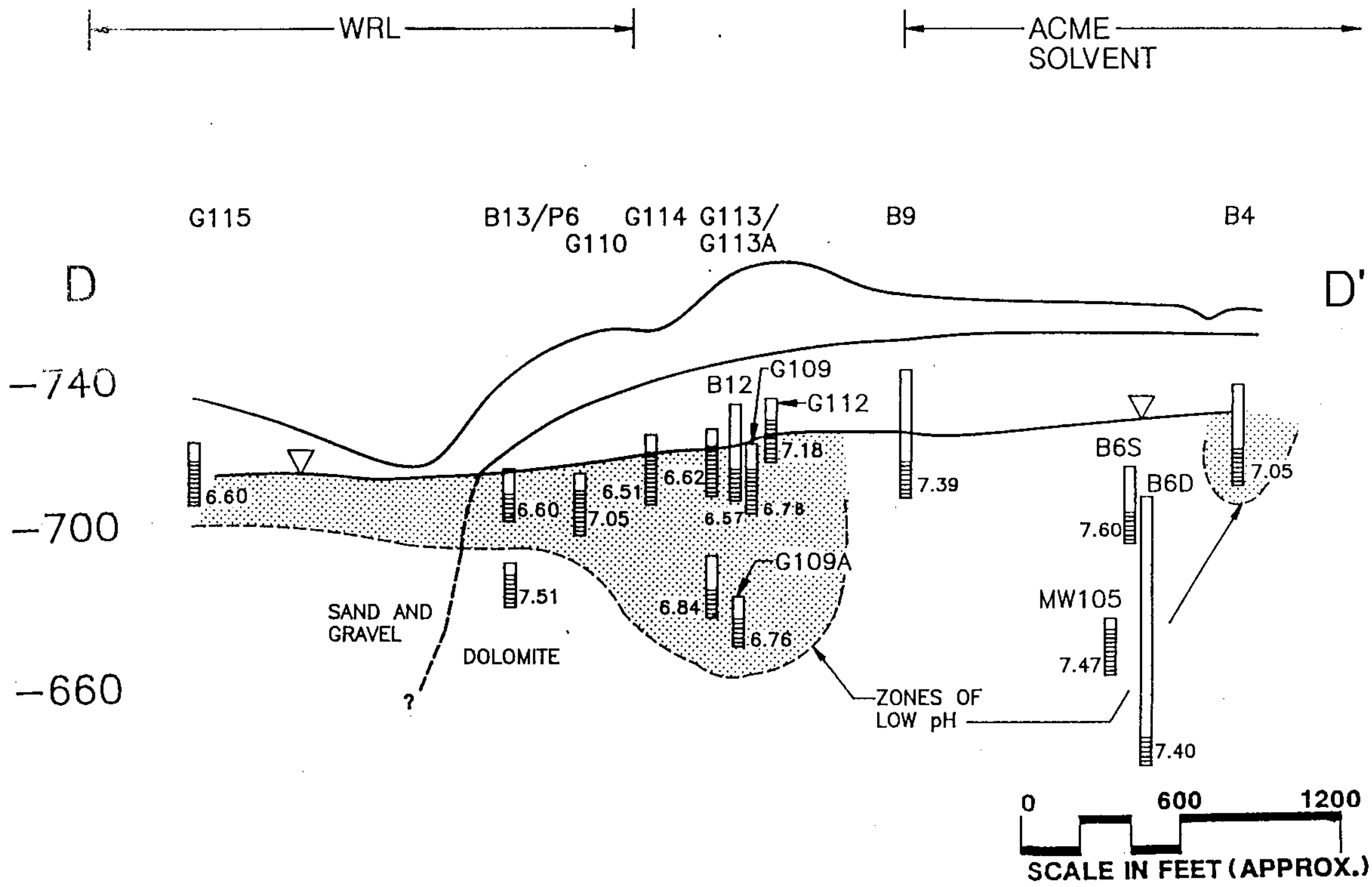


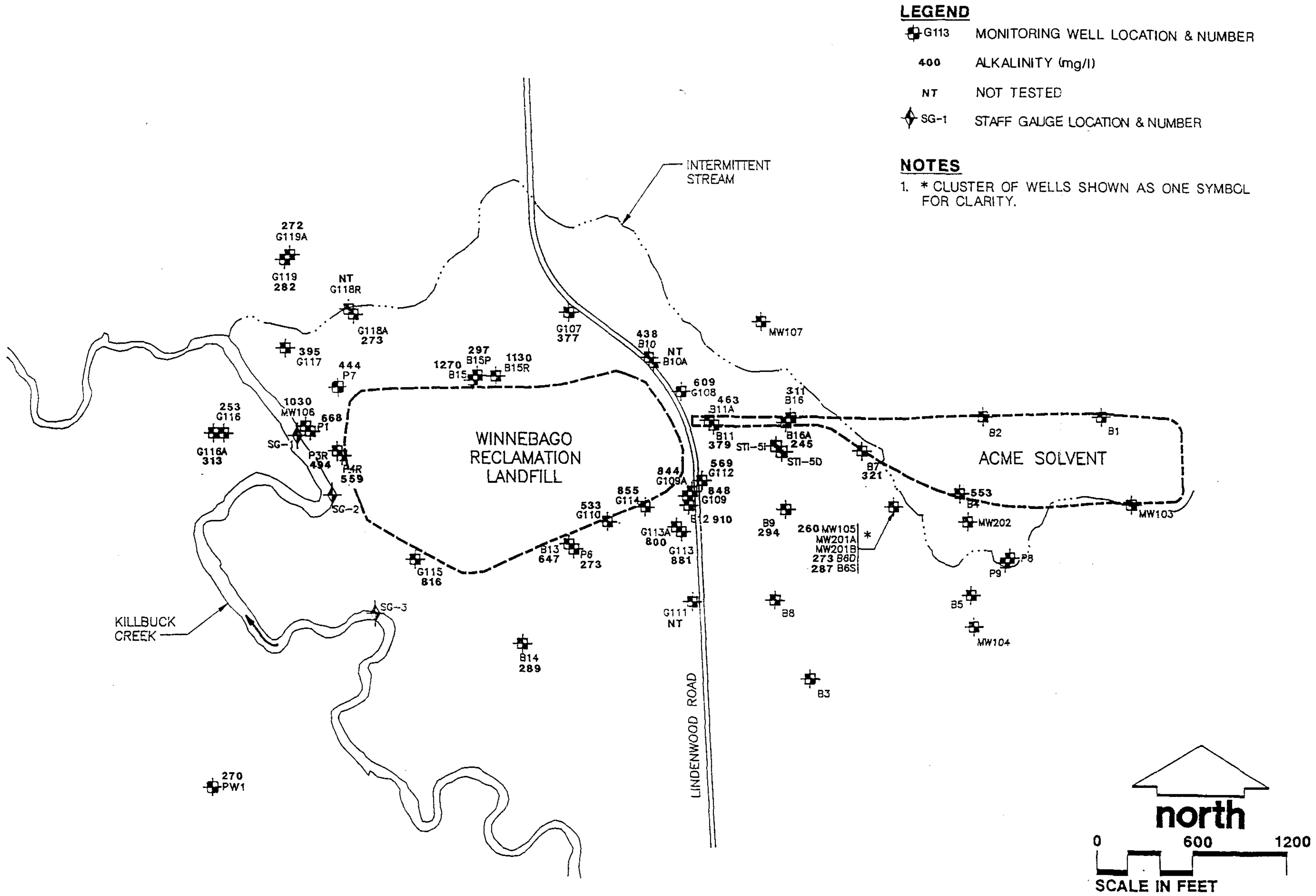
FIGURE 4-36

ROUND 2 pH DATA

WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS

WARZYN

13160 A21



ROUND 1 ALKALINITY

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

Revisions | Date | By | App'd

WARZYN
WARZYN ENGINEERING INC.

Designed By:
Drawn By: ELR, DLL.
Checked By: MJH
Approved By: Gary E. Parker
Date: 4/01/91
Reference:
File Path:

Project Number 13160 B26

WARZYN

FIGURE 4-34

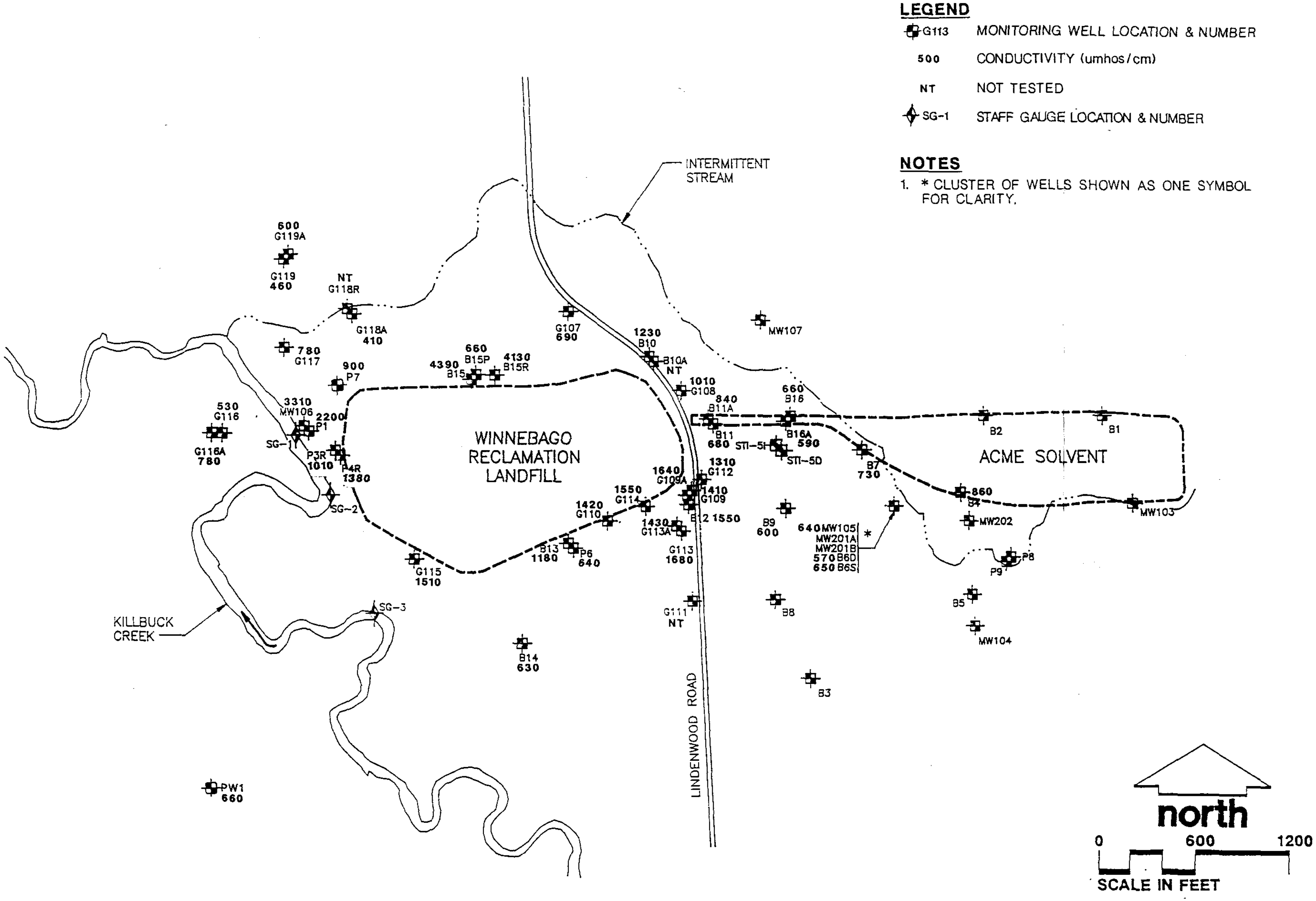


FIGURE 4-35

ROUND 1 SPECIFIC CONDUCTANCE

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

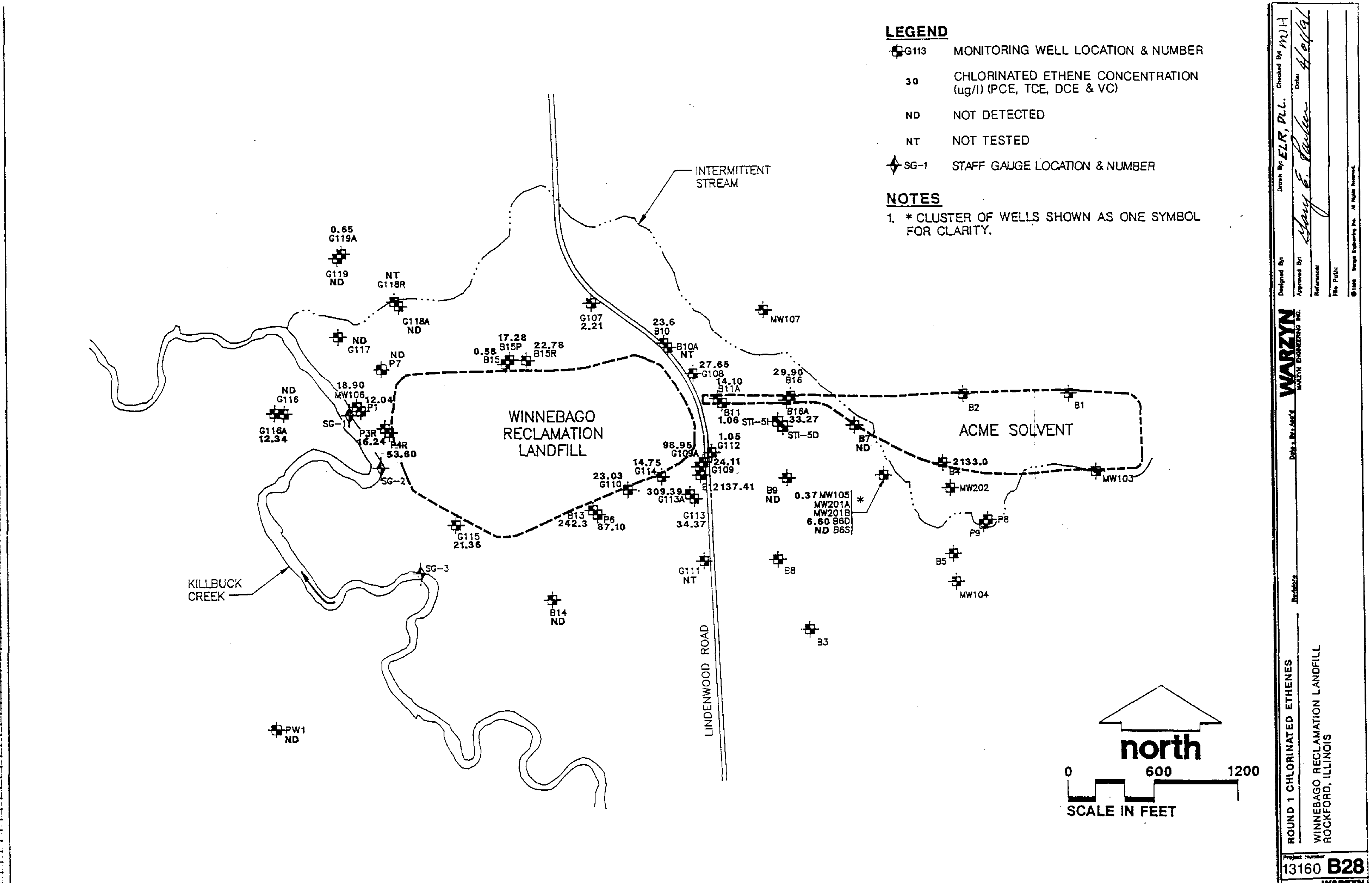
WARZYN
WARZYN ENGINEERING INC.

Revisions | Date | By | App'd

Designed By: | Drawn By: ELR, DLL. | Checked By: mJH
Approved By: [signature] | Date: 4/01/91
Reference:
File Path:

Project Number 13160 B27

WARZYN



LEGEND
G113 MONITORING WELL LOCATION & NUMBER
30 CHLORINATED ETHENE CONCENTRATION (ug/l) (PCE, TCE, DCE & VC)
ND NOT DETECTED
NT NOT TESTED
SG-1 STAFF GAUGE LOCATION & NUMBER
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
INTERMITTENT STREAM
WINNEBAGO RECLAMATION LANDFILL
ACME SOLVENT
KILLBUCK CREEK
LINDENWOOD ROAD
0.65 G119A
G119 ND
NT G118R
G118A ND
ND G117
ND P7
G107 2.21
23.6 B10
B10A NT
0.58 B15
17.28 B15P
22.78 B15R
MW107
27.65 G108
14.10 B11A
B11 1.06
29.90 B16
B16A
STI-5H 33.27
STI-5D
1.05 G112
98.95 G109A
24.11 G109
2137.41
14.75 G114
23.03 G110
309.39 G113A
G113 34.37
B13 242.3
P6 87.10
G115 21.36
ND G116
G116A 12.34
18.90 MW106
12.04 P1
P3R 16.24
P4R 53.60
SG-1
SG-2
SG-3
B14 ND
G111 NT
B9 ND
0.37 MW105
MW201A
MW201B
6.60 B6D
ND B6S
B8
B3
B7 ND
B2
B1
2133.0 B4
MW202
MW103
P8
P9
B5
MW104
PW1 ND
north
0 600 1200
SCALE IN FEET
ROUND 1 CHLORINATED ETHENES
WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS
13160 B28
WARZYN

FIGURE 4-38

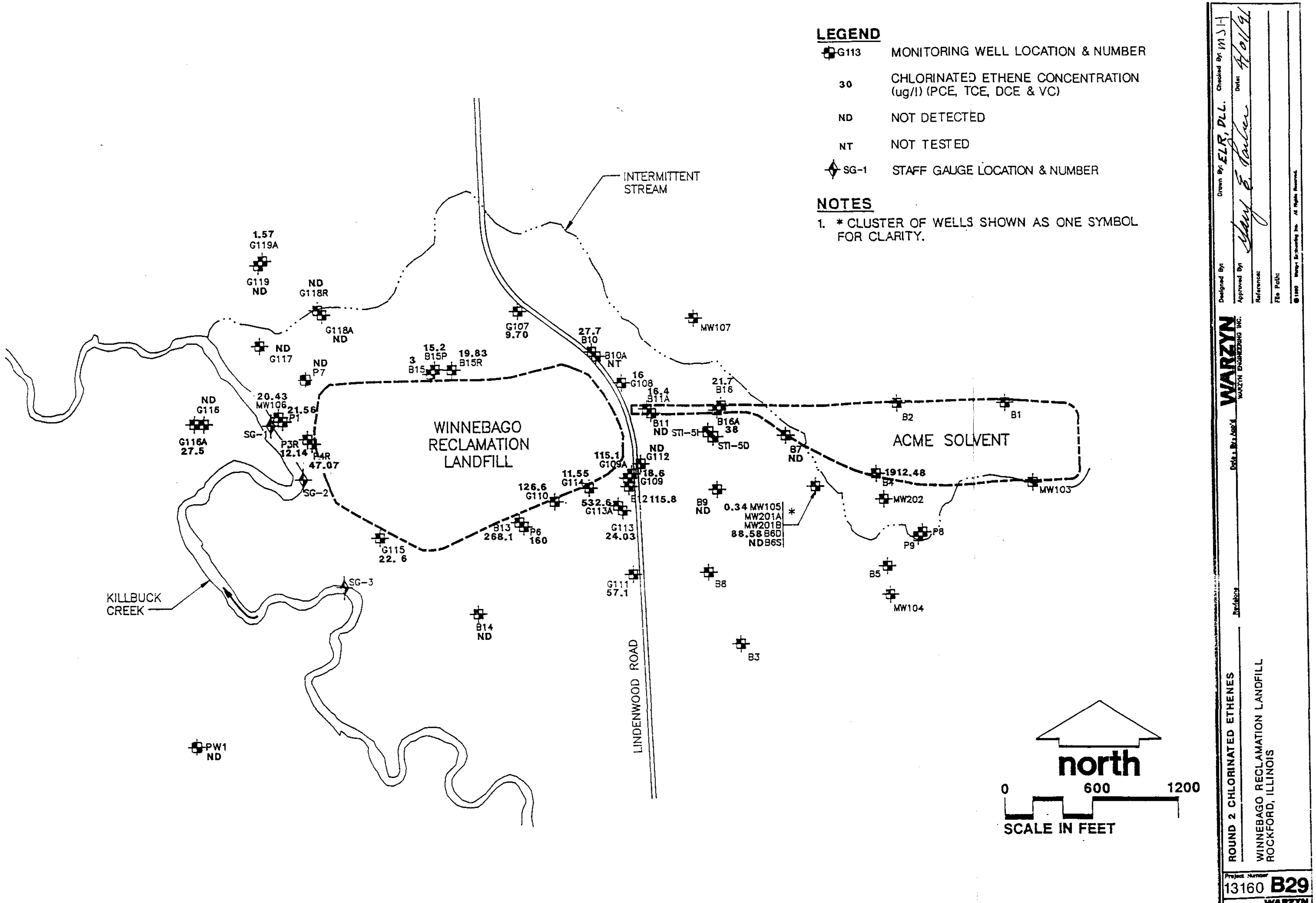
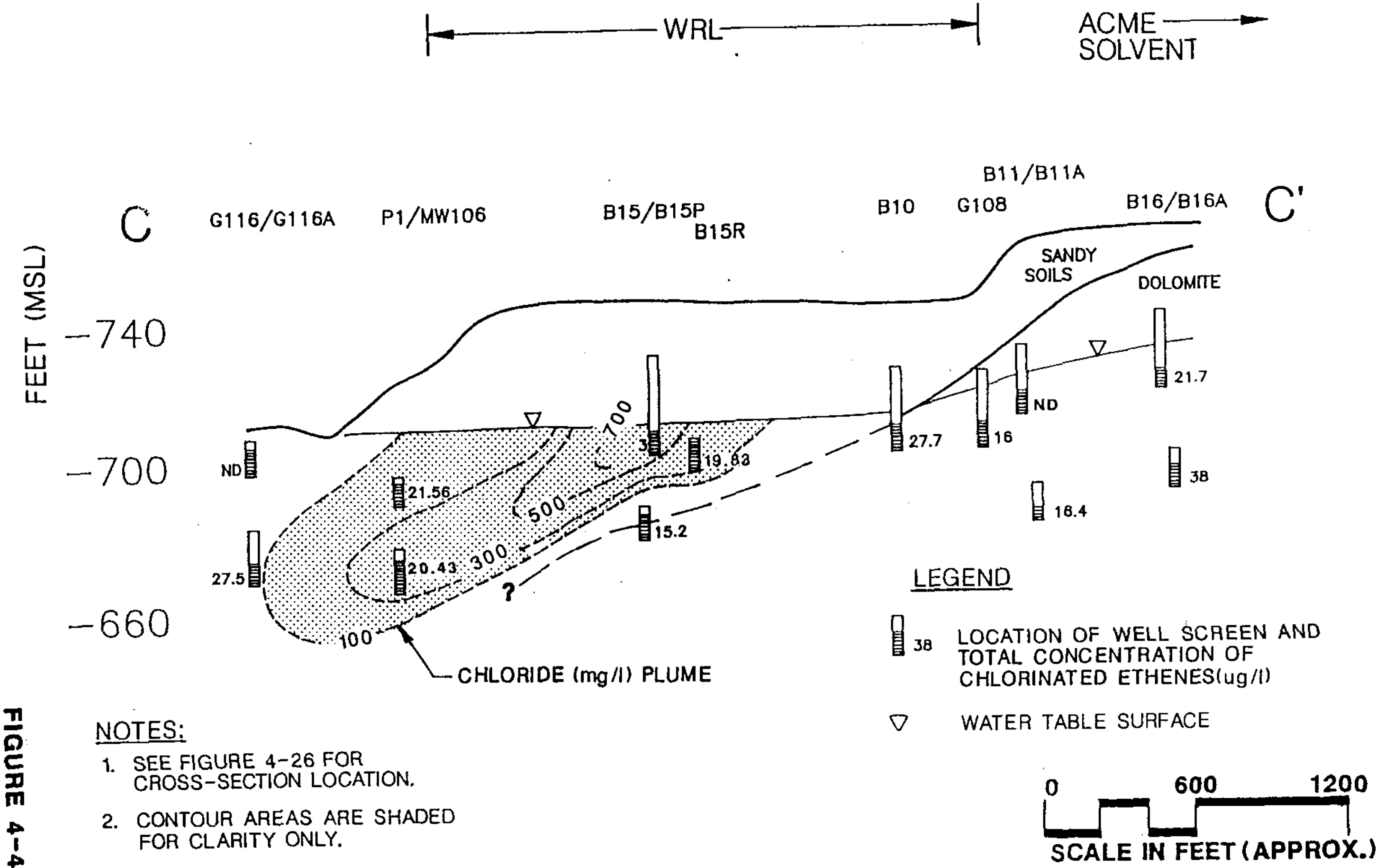

LEGEND
G113 MONITORING WELL LOCATION & NUMBER
30 CHLORINATED ETHENE CONCENTRATION (ug/l) (PCE, TCE, DCE & VC)
ND NOT DETECTED
NT NOT TESTED
SG-1 STAFF GAUGE LOCATION & NUMBER
NOTES
1. * CLUSTER OF WELLS SHOWN AS ONE SYMBOL FOR CLARITY.
INTERMITTENT STREAM
WINNEBAGO RECLAMATION LANDFILL
ACME SOLVENT
KILLBUCK CREEK
LINDENWOOD ROAD
north
0 600 1200
SCALE IN FEET
1.57 G119A
G119 ND
ND G118R
G118A ND
ND G117
ND P7
G107 9.70
27.7 B10
B10A NT
15.2 B15P
19.83 B15R
3 B15
16 G108
16.4 B11A
B11 ND
21.7 B16
B16A 38
STI-5H
STI-5D
MW107
B2
B1
B7 ND
1912.48 B4
MW202
MW103
P8
P9
B5
MW104
B3
B8
ND G116
G116A 27.5
20.43 MW106
21.56 P1
SG-1
P3R 12.14
P4R 47.07
SG-2
SG-3
115.1 G109A
ND G112
18.6 G109
B12 115.8
11.55 G114
126.6 G110
532.6 G113A
G113 24.03
B13 268.1
P6 160
G115 22.6
B9 ND
0.34 MW105
MW201A
MW201B
88.58 B6D
ND B6S
G111 57.1
B14 ND
PW1 ND
ROUND 2 CHLORINATED ETHENES
WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS
WARZYN
WARZYN ENGINEERING INC.
13160 B29

FIGURE 4-39



NOTES:

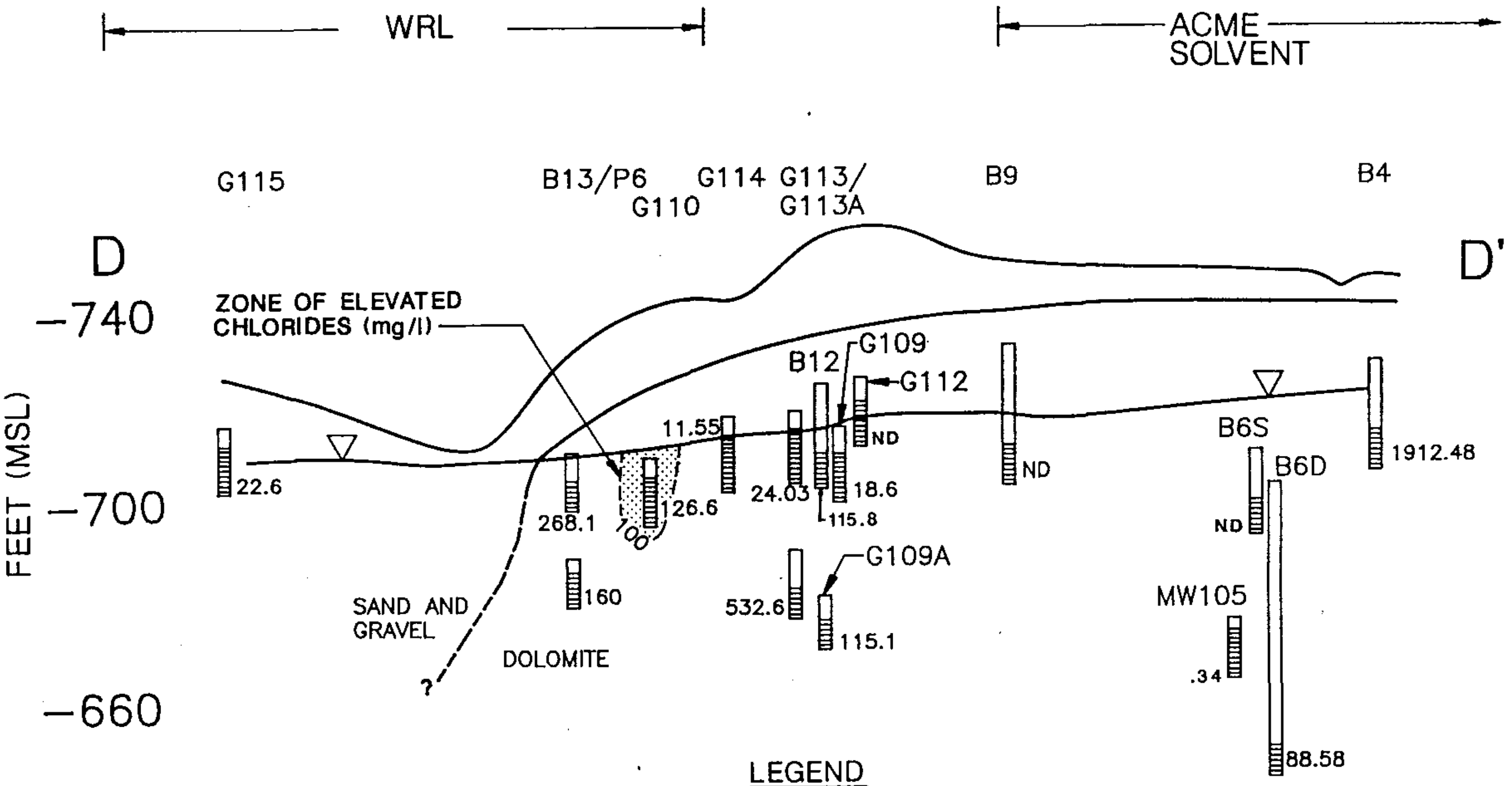
1. SEE FIGURE 4-26 FOR CROSS-SECTION LOCATION.
2. CONTOUR AREAS ARE SHADED FOR CLARITY ONLY.

WARZYN

FIGURE 4-40

ROUND 2 TOTAL ETHENES (ug/l)

WINNEBAGO RECLAMATION LANDFILL



NOTES:

1. SEE FIGURE 4-26 FOR CROSS-SECTION LOCATION.
2. CONTOUR AREAS ARE SHADED FOR CLARITY ONLY.

LEGEND

38 — LOCATION OF WELL SCREEN AND TOTAL CONCENTRATION OF CHLORINATED ETHENES (ug/l) (PCE, TCE, DCE & VC)

▽ — WATER TABLE SURFACE

0 600 1200
SCALE IN FEET (APPROX.)

FIGURE 4-41

WARZYN

ROUND 2 TOTAL ETHENES (ug/l)

WINNEBAGO RECLAMATION LANDFILL

NOTES:

1. SEE FIGURE 4-26 FOR CROSS-SECTION LOCATION.
2. CONTOUR AREAS ARE SHADED FOR CLARITY ONLY.

LEGEND

38 LOCATION OF WELL SCREEN AND TOTAL CONCENTRATION OF ALL VOC's (ug/l)

▽ WATER TABLE SURFACE

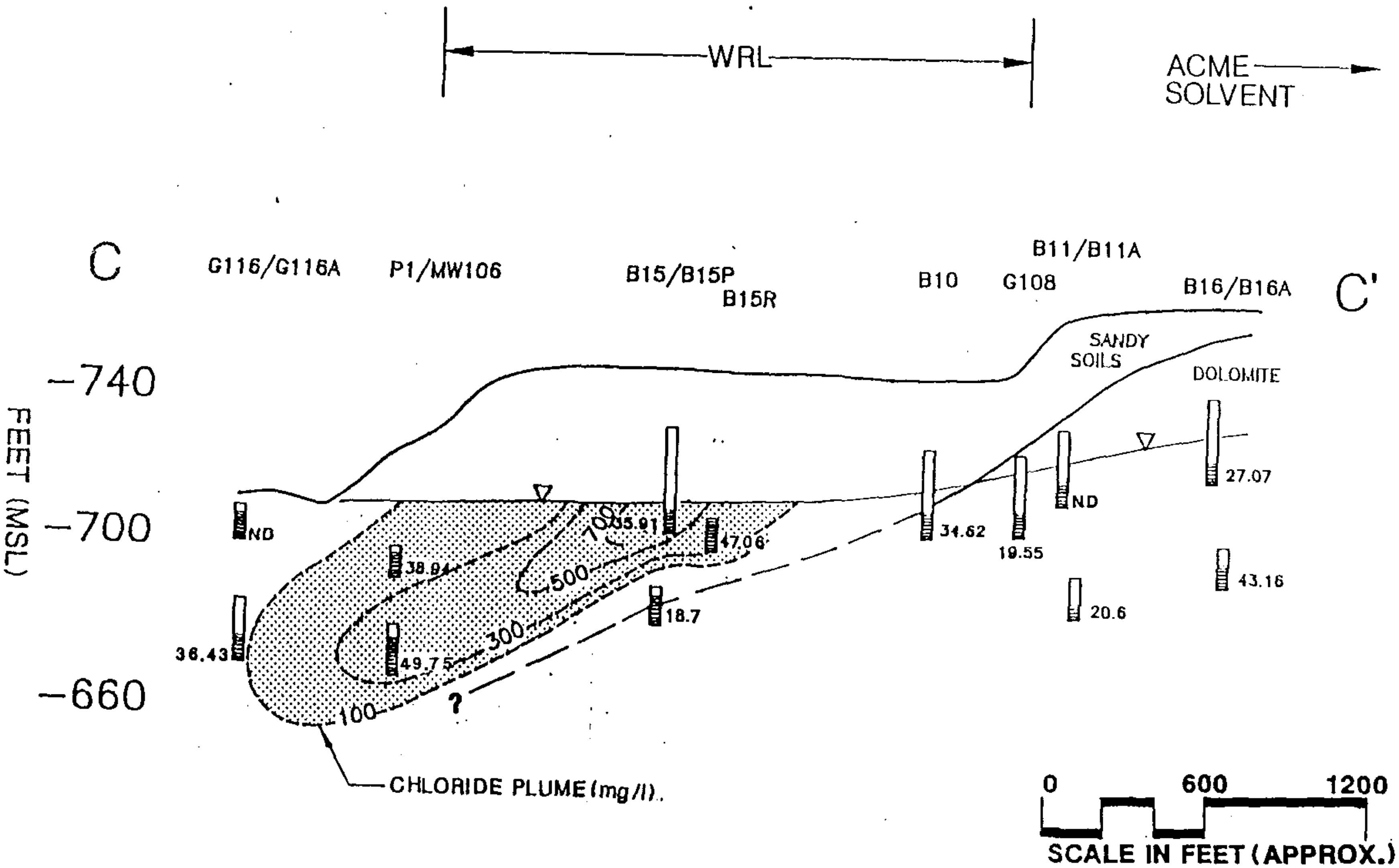


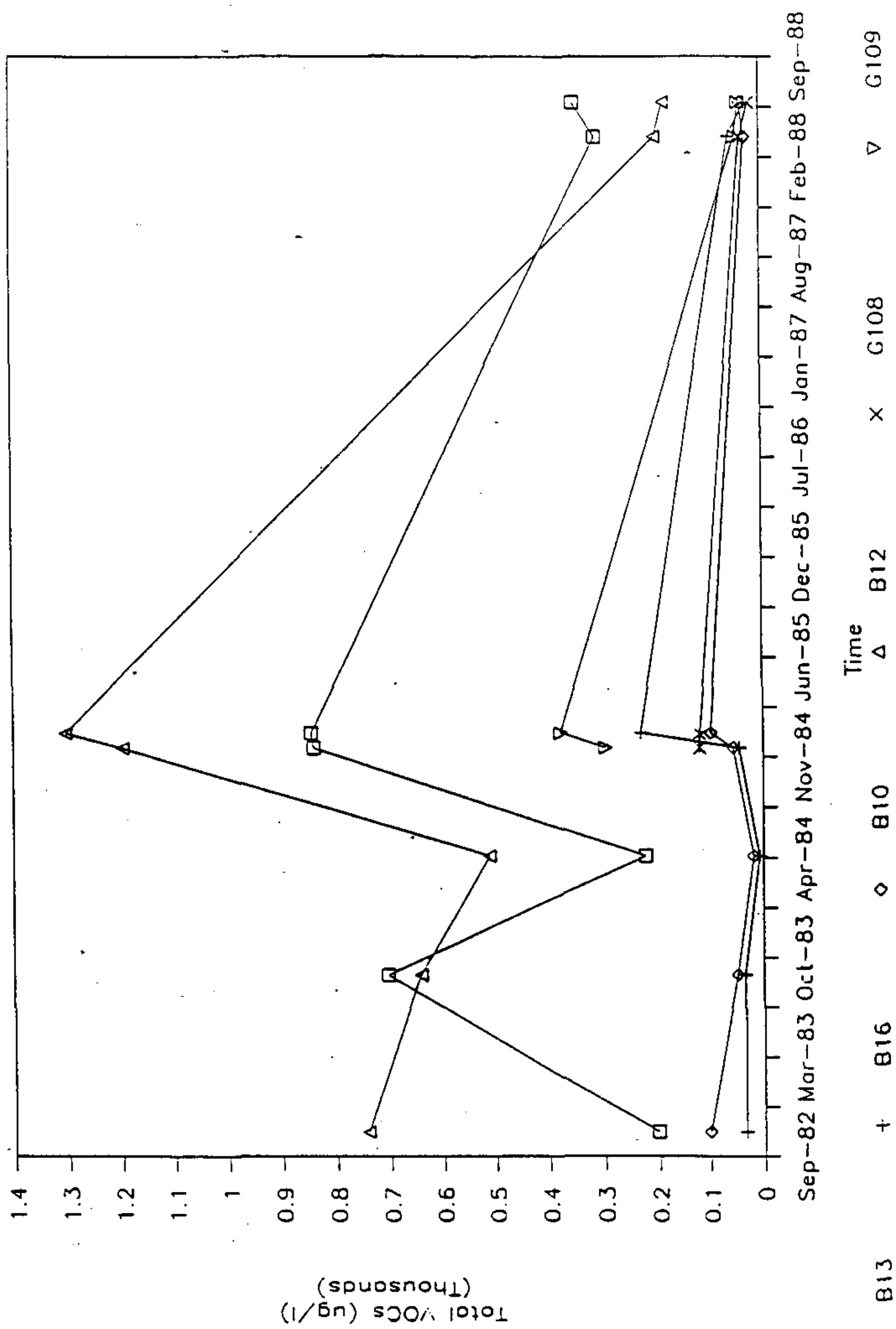
FIGURE 4-42

ROUND 2 TOTAL VOCs (ug/l)

WINNEBAGO RECLAMATION LANDFILL
ROCKFORD, ILLINOIS

WARZYN

DWN ELR
DATE 11-22-?
13160
A25



NOTES

1. TOTAL VOCs VALUES DO NOT INCLUDE METHYLENE CHLORIDE OR ACETONE CONCENTRATIONS.
2. OCTOBER & SEPTEMBER 1983 SAMPLING EVENTS CONDUCTED BY ECOLOGY AND ENVIRONMENT, INC. (1983).
3. DECEMBER 1984, JANUARY 1985, APRIL 1988 & JUNE 1988 SAMPLING EVENTS CONDUCTED BY WARZYN INC. (1985) AND (1990).
4. MAY 1984 SAMPLING EVENT CONDUCTED BY THE E.C. JORDAN COMPANY (1984).

FIGURE 4-43

WARZYN	TOTAL VOC's(ug/l) OVER TIME	Drawn DLL	Checked MJH	App'd. 4EP
	WINNEBAGO RECLAMATION LANDFILL ROCKFORD, ILLINOIS			Date 4-01-91
				13160 **A26**

SECTION 5
CONTAMINATION FATE AND TRANSPORT

5.1 Introduction

The fate and transport of contaminant compounds identified at, or adjacent to, the WRL Site are dependent on several factors, including:

- Chemical and physical characteristics of the contaminants
- Characteristics of the contaminated media (e.g., aquifer, or soil type, organic carbon content)
- Source of contamination
- Climatic conditions
- Potential migration pathways

Each of the above factors, as they relate to the WRL Site, will be discussed in this section. Inclusion of compounds in the following discussion should not necessarily be construed as the WRL Site being the source of the compounds.

5.2 Main Sources of Contamination and Transport Pathways

There are three general areas where chloride concentrations are elevated, indicating that WRL leachate is a source of contaminants to the groundwater. These are:

- The northwest quadrant of the WRL Site as defined by wells B15R, MW106, P1, P4R and G116A
- In the vicinity of wells G110 and G114
- In the vicinity of well G115

The groundwater upgradient of the WRL Site has been impacted by VOCs with the highest concentrations being observed at well B4, indicating that an upgradient source of VOCs to the groundwater exists. Although migration of landfill gas as a source of VOCs to the groundwater upgradient of the WRL Site cannot be completely ruled out, it is not likely as discussed in Section 4.7.2. The potential of nearby septic leaching fields to affect groundwater quality also cannot be ruled out based upon available data. Regardless of the source, groundwater flow is believed to be the major pathway by which contaminants are transported in the vicinity of the WRL Site.

Surface water, sediment, and air are other possible media which may transport contaminants, but are of lesser importance based on current WRL Site conditions. Groundwater and surface water flow and contaminant distribution within each media are presented in detail in previous sections of this report. The behavior (fate and transport) of the identified chemical contaminants within the environment surrounding the WRL Site is the focus of this section.

5.3 Factors Affecting Contaminant Fate and Transport

Contaminant fate and transport is largely dependent on specific physical and chemical properties, such as water solubility, specific gravity, vapor pressure, chemical and biological degradability, organic carbon partitioning coefficient (Koc), soil/water partitioning coefficient (K_d), and Henry's Law constant. The mechanisms of fate and transport will be discussed in relation to how they apply to specific groups of contaminants that were identified during the RI. Howard (1990) and Dragun (1988) were reviewed and served as a basis for discussions presented in Sections 5.3.1 and 5.3.2. The contaminant groupings are based on similarities in how certain classes of chemicals behave in media. The classes of chemicals which were detected on-Site can be segregated into a number of subclasses based on their similar chemical characteristics and uses. Not all chemicals detected within a subclass of chemicals at the WRL Site are discussed. The chemicals listed under each subclass are for example purposes, only. Refer to Section 6.2.3.1 for a list of potential contaminants of concern, and Table 6-9 for chemical and physical properties relevant to fate and transport.

5.3.1 Volatile Organic Chemicals (VOCs)

As their name implies, volatile organic chemicals easily evaporate, and can exist in both the dissolved and vapor states. These compounds are generally more water soluble than the other classes of chemicals (e.g., PCBs), do not adsorb as readily to soils or aquifer materials, and are biologically degraded more easily than some heavier molecular weight compounds (e.g., PAHs). Due to their relatively higher solubilities, VOCs are more readily transported by groundwater than other types of organic compounds. Of the VOCs listed, the BETX compounds (i.e., non-chlorinated single benzene ring chemicals) are more easily degraded than the chlorinated solvents (e.g., chlorinated ethenes) under aerobic conditions. The biological degradation of the chlorinated VOCs under anaerobic conditions has been discussed in Section 4.0. The chlorinated VOCs may also be subject to hydrolysis reactions in the groundwater system. VOCs would be expected to be somewhat to highly mobile in the soil environment, and subject to migration within groundwater, as well as vapor transport due to volatilization.

The following is a fate and transport discussion for the major subclasses of VOCs detected adjacent to the WRL Site. Descriptions regarding volatility and water solubility are based on relative comparisons of vapor pressures and solubilities for individual compounds. The terminology regarding mobility is based on values for the organic carbon partition coefficient as follows (Dragun, 1988-Table 6.6):

Koc	Mobility Class
>2000	Immobile
500-2000	Relatively Immobile
150-500	Relatively Mobile
50-150	Mobile
<50	Very Mobile

- <u>BETX Compounds</u> - BETX is an acronym representing the compounds benzene, ethylbenzene, toluene, and xylene. BETX compounds are typically used as solvents and are derived from oil and/or hydrocarbon products.

  All four compounds can exist in the dissolved and soil vapor phases based on their vapor pressures. Ethylbenzene is slightly less volatile than the other three. The BETX compounds are relatively soluble in water and would be expected to be present in aqueous leachate and groundwater phases. Based on their organic carbon partitioning coefficients, benzene would be expected to be mobile, xylene and toluene relatively mobile, and ethylbenzene relatively immobile in both soil and groundwater environments. The BETX compounds are subject to biodegradation under aerobic and anaerobic conditions. The biodegradation of aromatic hydrocarbons can occur under the anaerobic conditions which may be present within the landfill and shallow aquifer. Intermediates of anaerobic biodegradation of aromatic hydrocarbons include derivatives of cyclohexane and aliphatic hydrocarbons. Hydrolysis within the groundwater phase is not expected to be a degradation pathway for this classification of compounds. Hydrolysis of aerobic biodegradation intermediates may produce various benzene derivatives.

- Chlorinated Ethenes - Chlorinated ethenes include trichloroethene, tetrachloroethene, cis- and trans-1,2-dichloroethene, and vinyl chloride. Trichloroethene and tetrachloroethene are used as solvents and degreasing agents, while cis- and trans-1,2-dichloroethene and vinyl chloride usually represent anaerobic biodegradation products of other chlorinated compounds. Vinyl chloride also is used as a monomer in the manufacture of plastics.

  All five compounds can exist in the dissolved and vapor phases based on their vapor pressures. Vinyl chloride and cis- and trans-1,2-dichloroethene are extremely volatile, while tetrachloroethene is only moderately volatile. The chlorinated ethenes are relatively soluble in water and would be expected to be present in aqueous leachate and groundwater phases. Tetrachloroethene is somewhat less soluble than the other three. Based on their organic carbon partitioning coefficients, vinyl chloride, cis- and trans-1,2-dichloroethene, and trichloroethene would be expected to be mobile to very mobile and tetrachloroethene relatively mobile in both soil and groundwater environments. The chlorinated ethenes are subject to biodegradation under anaerobic conditions which may be present within the landfill and shallow aquifer. Vinyl chloride typically is biodegradable under aerobic conditions, only, and has a very long biodegradation half-life. Hydrolysis of chlorinated ethenes within the groundwater phase is a possible degradation pathway, but reaction rates should be extremely slow.

- Chlorinated Ethanes - Chlorinated ethanes include 1,1,2,2-tetrachloroethane, 1,1,2-trichloroethane, 1,1,1-trichloroethane, 1,1-and 1,2-dichloroethane, and chloroethane. These compounds are used as solvents and degreasing agents and may also represent an anaerobic biodegradation sequence.

  The chlorinated ethanes can exist in the dissolved and vapor phases based on their vapor pressures. 1,1,2,2-tetrachloroethane is only moderately volatile. The chlorinated ethanes are relatively soluble in water and would be expected to be present in aqueous leachate and groundwater phases. Based on their organic carbon partition coefficients, the chlorinated ethanes would be expected to be mobile to very mobile in soil and groundwater environments. The chlorinated ethanes are subject to biodegradation under anaerobic conditions which may be present within the landfill and shallow aquifer. Hydrolysis of chlorinated ethanes within the groundwater phase is a possible degradation pathway, but reaction rates should be extremely slow.

- Chlorinated Methanes - The chlorinated methanes include chloromethane, methylene chloride, chloroform, and carbon tetrachloride. Methylene chloride may be present as a laboratory artifact. These compounds are used as solvents and degreasing agents. Fate and transport mechanisms for chlorinated methanes are similar to chlorinated ethanes discussed above. Carbon tetrachloride is only moderately volatile and has a lower solubility in water than the other chlorinated methanes.

- Chlorinated propanes and propenes - Chlorinated propane and propene compounds of potential concern at the WRL Site include 1,3-dichloropropene and 1,2-dichloropropane. These compounds are used as solvents, as well as a soil fumigant and intermediates in tetrachloroethene and carbon tetrachloride production.

  Both of these compounds can exist in the dissolved and vapor phases based on their vapor pressures. They are less volatile than the chlorinated compounds previously discussed, but similar to the BETX compounds. Both compounds are relatively soluble in water and would be expected to be present in aqueous leachate and groundwater phases. Based on their organic carbon partitioning coefficients, both would be expected to be very mobile in both soil and groundwater environments. The chlorinated propanes and propenes may be subject to biodegradation under both aerobic and anaerobic conditions. Hydrolysis of chlorinated propanes and propenes in the groundwater phase is a possible degradation pathway, but reaction rates should be extremely slow.

- Ketones - Based on arguments made in Section 6.0 (Risk Assessment), acetone is the only compound included in this classification. Acetone may be present as a laboratory or sampling artifact. Acetone is used as a solvent and as an intermediate in the production of other organic chemicals.

  Acetone can exist in the dissolved and vapor phases based on its vapor pressure. It is extremely soluble in water and would be expected to be present in aqueous leachate and groundwater phases. Based on its organic carbon partitioning coefficient, acetone would be expected to be very mobile in both soil and groundwater environments. Acetone is readily degraded under aerobic soil and groundwater conditions, but would not be expected to degrade rapidly under anaerobic conditions which may be present within the landfill and shallow aquifer. Hydrolysis of acetone in the groundwater phase is a possible degradation pathway, but reaction rates should be extremely slow.

5.3.2 Semi-Volatile Organic Compounds (SVOCs)

As their name implies, SVOCs are less volatile than VOCs, and therefore are not as readily present in the vapor phase. This class of compounds is generally less water soluble than VOCs, and adsorb more readily to soils or aquifer materials. These compounds are preferentially attracted to soil or aquifer materials due to their hydrophobic (i.e., water disliking) nature. In general, the soil components that have the greatest effect on the adsorption of SVOCs to soil and aquifer materials are organic matter, clay and hydrous oxides. A soil system's adsorption capacity is typically correlated to its organic carbon content. These organic and clay materials with their large amount of surface area tend to preferentially attract hydrophobic chemicals. Adsorptive forces greatly influence and limit the mobility of SVOCs in soil and aquifer systems. Because the water solubility of this class of chemicals varies substantially (e.g., chlorinated benzenes vs. high molecular weight polynuclear aromatic hydrocarbons), the mobility and degradability of individual compounds varies greatly within the group. The more complex polynuclear aromatic hydrocarbons (e.g., acenaphthylene) are many orders of magnitude less water soluble than chlorinated benzenes. Because of differences in water solubilities, chlorinated benzenes would be more likely to be present in groundwater than polynuclear aromatic hydrocarbons and phthalates. When SVOCs are bound to surface soils, the main transport mechanisms are sediment transport by run off or erosion and/or particulate transport by wind or erosion.

The following are the major subclasses of SVOCs detected at the WRL Site:

- Chlorinated Benzenes - Chlorinated benzenes include 1,2-dichlorobenzene, 1,4-dichlorobenzene, and chlorobenzene. These compounds are used as fumigants, pesticides, and dye intermediates. 1,4-dichlorobenzene has been used as a main component of bathroom deodorizers, toilet bowl cleaners, and moth balls.

  The chlorinated benzenes can exist in the dissolved and vapor phases based on their vapor pressures. Chlorobenzene is more volatile than the other two chlorinated benzenes detected. Its volatility is similar to BETX compounds. The dichlorobenzene compounds are less volatile than the VOCs discussed in Section 5.3.1. The chlorinated benzenes are relatively soluble in water and would be expected to be present in aqueous leachate and groundwater phases. Based on their organic carbon partitioning coefficients, chlorobenzene would be expected to be relatively mobile and the dichlorobenzene compounds relatively immobile in both soil and groundwater environments. The chlorinated benzene compounds are subject to biodegradation under aerobic soil and groundwater conditions, but would not be expected to degrade rapidly under anaerobic conditions which may be present within the landfill and shallow aquifer. Hydrolysis within the groundwater phase is not expected to be a degradation pathway for this classification of compounds.

- Phthalates - Based on arguments made in Section 6.0 (Risk Assessment), bis(2-ethylhexyl) phthalate is the only compound included in this classification. Phthalates are used as plasticizing agents, but may also be naturally occurring.

Phthalates would be expected to exist primarily in the adsorbed soil phase, although because of their limited solubility in water, they could be present in aqueous leachate and groundwater phases. Based on their organic carbon partitioning coefficients, they would be expected to be immobile in both soil and groundwater environments. Phthalates are subject to biodegradation under aerobic soil and groundwater conditions, but are likely to have long biodegradation half-lives. They would not be expected to degrade rapidly under anaerobic conditions which may be present within the landfill and shallow aquifer. Hydrolysis within the groundwater phase is not expected to be a degradation pathway for this classification of compounds.

- Polycyclic Aromatic Hydrocarbons (PAHs) - PAHs include a group of compounds derived from coal tars and oil, including naphthalene, pyrene, acenaphthylene, etc.

  PAHs would be expected to exist primarily in the adsorbed soil phase. However, like phthalates, because of their limited solubility in water, they could be present in aqueous leachate and groundwater phases. Napthalene has the highest vapor pressure and water solubility of the PAHs, and is most likely to be present in the vapor and aqueous phase. Based on their organic carbon partitioning coefficients, PAHs would be expected to be immobile in both soil and groundwater environments. Because of their large molecular structure, PAHs do not readily biodegrade under normal soil and groundwater conditions. Under the right conditions, PAHs can be aerobically degraded. They would not be expected to degrade under anaerobic conditions which may be present within the landfill and shallow aquifer. Hydrolysis within the groundwater phase is not expected to be a degradation pathway for this classification of compounds.

5.3.3 PCBs/Pesticides

The chemistry of pesticides and PCBs is generally similar in nature to the SVOCs. This class of chemicals has a low solubility in water and is generally high in molecular weight. Unlike the SVOCs listed above, the pesticides detected in the WRL Site leachate are highly chlorinated. As mentioned before, chlorinated compounds degrade less readily than non-chlorinated chemicals under aerobic conditions. Due to their large molecular structure, the pesticides and PCBs would tend to be similar in mobility to the higher molecular weight SVOCs, but would not degrade as readily.

The following are the major subclasses of PCBs/Pesticides detected at the WRL Site:

- PCBs - High molecular weight variably chlorinated biphenyls which were formerly used as industrial chemicals (e.g., in hydraulic fluids) and also associated with capacitors and transformers for electronic products (e.g., Arochlor 1248)

- Pesticides - High molecular weight chlorinated compounds (e.g., Aldrin and Endrin) which were popular in the 1960's and early 1970's, but which generally are not currently used in the United States except for special permitted uses

5.3.4 Inorganics and Cyanide

The inorganic constituents analyzed for at the WRL Site included nitrate-nitrite nitrogen, sulfates, metals, cyanide, and chloride. In general, metals are positively charged (cations). Chloride is non-metallic and negatively charged (anion), and although cyanide is not truly an inorganic species, it behaves like an anion and therefore its chemistry is more similar to that of inorganic anions than organic compounds.

Metals typically exist in the soil environment in the form of oxides, hydroxides, and carbonates. Because of their cationic nature, metals are bound readily to negatively charged surfaces contained on organic matter, clay, and hydrous metal oxides contained in dusts, sediment, soil, and aquifer material (i.e., opposite charges attract), thus limiting their mobility in these media. The amount of these reactive components determines the attenuation capacity of the material. For example, surface soil and sediments surrounding the landfill would be expected to have more attenuation capacity than the sand and gravel or dolomite mineral of which the aquifers are composed. The soil and sediment would be expected to have a moderate quantity of organic matter and clay, while the aquifer materials probably contain only lesser amounts of these constituents. Therefore, sediment and fine grained surface soils would be able to retain more metals and prevent their migration more effectively than the dolomite or sand and gravel aquifers. Anions, such as chloride, are not attracted (retained) as readily as cations to aquifer and soil materials, because of the low amount of positive charge on their surfaces. As discussed in Section 3, chloride is a non-reactive species in groundwater systems and generally only dilution accounts for its attenuation.

Many metals are generally insoluble or sparingly soluble in groundwater within the normal pH range of groundwater (6 to 8), and therefore are precipitated as metallic salts. However, there is wide variability in the solubility of metals depending on the particular salts of the metal which are present in the soil or aquifer materials. Usually within the soil or aquifer system a select few insoluble species of each metal control the solubility of those metals within the soil/groundwater environment. Among metals, the more alkaline metals such as sodium, calcium, potassium, and magnesium tend to be most soluble in water within the normal pH of groundwater and soil pore water.

The oxidation/reduction potential (pE) decreases as oxygen is depleted from the soil environment, which causes reducing (i.e., less oxidizing) conditions to occur. In relation to the landfill, if biologically active leachate enters the groundwater, biological degradation of the nutrient rich leachate may reduce the oxygen in the groundwater and decrease the pE. As the pE decreases, the solubility of metals may increase. Therefore, under low pE conditions, some of the metals which are generally adsorbed readily to soil and aquifer materials may be more soluble in groundwater or pore water and thus more readily transported. However, as the pH and pE conditions return to normal during transport, the metals become attenuated by the soil and aquifer materials. In general, metals are not volatile species, and therefore are not readily transported in air under ambient conditions unless surface soil is eroded by wind and causes dust generation.

The soil chemistry of metals can be divided into two categories. These categories include:

- Solution chemistry
- Interfacial chemistry

Solution chemistry involves the fixation of the metal directly into the soil structural complex. Fixation refers to the soil chemical reactions which immobilize an element within the structure of a mineral. Since soil minerals are made up of naturally occurring metal complexes, interactions between contaminant metals and mineral complexes can result in preferential metal exchange. Contaminant metals can be preferentially exchanged for naturally occurring metals present in the soil mineral crystal latices. Fixation of a metal into the mineral complex is the most effective mechanism of metals attenuation in soils.

As mentioned previously, fixation of metals into the soil complex is dependent on the oxidation/reduction potential (pE) and pH of the soil medium and the solubility of the individual metal complexes that are prevalent under those conditions. Metal ions also form many soluble complexes with both organic and inorganic ligands. The effect of metal complexation with ligands is to increase the solubility, and thus mobility, of the metal complex.

Interfacial chemistry involves the cation exchange interactions between metal cations and anionic groups located on surface of soil media. These cation exchange interactions result in the adsorption of metals to the mineral surface. The negative charge of organic matter is pH dependent. Cations are held loosely in the vicinity of these negatively charged sites by electrostatic forces. Unlike fixation of metals into mineral complexes, cation exchange sites are non-specific. Adsorption is a less effective mechanism of metal attenuation since cations are readily exchanged at the mineral surface. Mobility of metals can be increased if the cation exchange capacity of the soil becomes saturated, or if the pH or oxidation/reduction potential of the soil environment changes.

The following general rules-of-thumb exist for adsorption interactions between metals and mineral surfaces:

- The higher the valence state of a cation, the greater its attraction to soils
- Larger cations are preferentially adsorbed
- Cation adsorption is significant at pH>6
- Cation adsorption increases with pH

The following is a brief summary of the soil chemistry for metals being evaluated in the WRL risk assessment. The information presented in this section is based on a review of Dragun (1988) and Sims et al. (1984):

<u>Arsenic</u> - Arsenic may exist in the +3 (arsenite) or +5 (arsenate) valence states. Arsenite is the more toxic form.

Arsenate may form insoluble precipitates with iron (e.g., $FeAsO_4$), aluminum and calcium, and can be leached if these compounds are present at low levels in the soil medium. Iron presence often controls the mobility of arsenate. Maximum arsenate adsorption occurs at a pH of 5; adsorption then decreases with increasing pH.

Arsenite may be predominant under anaerobic conditions which may exist in the landfill and shallow aquifer. Arsenite is 4 to 10 times more mobile than arsenate. Adsorption of arsenite increases with pH. Under anaerobic conditions, arsenite could exist as the volatile gas AsH_3. High organic matter, warm temperatures, adequate moisture and other conditions conducive to microbial activity drive the reaction toward methylation and volatilization which reduces arsenic residues in soil. An experimental study (Woolson, 1977) showed that only 1 to 2 percent of arsenate applied at 10 ppm was volatilized as arsine in 160 days.

Barium - Barium exists in the +2 valence state and may form the complexes $Ba(OH)_2$, $Ba_3(PO_4)_2$ and $BaSO_4$. Barium does not readily form complexes and often exists in its cation state. Adsorption would be the primary attenuation mechanism for the barium cation.

Cadmium - Cadmium exists in the +2 valence state and may form the complexes $Cd(OH)_2$, $CdCO_3$ and $Cd_3(PO_4)_2$. Cadmium's soil chemistry is similar to that of lead, except that cadmium compounds are more soluble than corresponding lead compounds at any given pH. Experimental studies have shown cadmium to be less mobile in the soil environment than nickel, lead, cobalt, and zinc.

Cobalt - Cobalt exists in the +2 valence state. It can be highly mobile in the soil environment due to its ability to complex with water soluble soil organic matter.

Lead - Lead exists in the +2 valence state and may form the complexes $Pb(CO_3)$, $Pb(OH)_2$, $Pb_3(PO_4)_2$ and $Pb_5(PO_4)_3Cl$. All of these precipitates may exist simultaneously around a pH of 7.5 to 8.0. As with other metal compounds, the solubility of lead compounds decrease with increasing pH. The solubility of lead increases again at a pH of 11 or greater. Experimental studies have shown lead to be more mobile than cadmium and zinc, but less mobile than nickel, in the soil environment.

Manganese - Manganese exists in the +2, +3 and +4 valence states and may form the complex $MnCO_3$ or various manganese hydroxides. The +2 valence state predominates in most natural water situations. Organic complexes may play a role in its transport in some situations (Hem, 1989)

Nickel - Nickel exists in the +2 valence state and may form the complexes $Ni(OH)_2$ and $NiFe_2O_4$. Experimental studies show nickel to be more mobile in the soil environment than lead, cobalt, zinc, and cadmium. It can be highly mobile in the soil environment due to its ability to complex with water soluble soil organic matter. Under anaerobic conditions which may exist in the landfill and the shallow aquifer, nickel may exist in its elemental form. (i.e., valence state of 0).

Silver - Silver exists in the +1 valence state and may form the complexes AgCl, Ag_2SO_4, $Ag(OH)_2$ and Ag_2CO_3. Precipitates of silver are highly insoluble. Therefore, silver is highly immobile in the soil environment. Under anaerobic conditions which may exist in the landfill and shallow aquifer, silver may exist in its elemental form (i.e., valence state of 0).

Zinc - Zinc exists in the +2 valence state and may form the complexes $Zn(OH)_2$, $ZnCO_3$ and $ZnSiO_4$. Experimental studies have shown zinc to be less mobile than nickel, lead, and cobalt and more mobile than cadmium in the soil environment.

5.4 Media Specific Mechanism of Fate and Transport

Table 6-1 summarizes the occurrence of the majority of the chemicals detected at the WRL Site in the media evaluated. Based on Table 6-1, it is apparent that VOCs and metals are the most prevalent groups of contaminants of concern detected within the media investigated (i.e., leachate and groundwater). Which properties will determine how mobile a contaminant will be is dependent to a large degree on the chemical's characteristics and the characteristics of the media which the chemical has contaminated. The distribution of contaminants in the groundwater and surface water body (i.e., surface water and sediment) will be explained in a qualitative manner. The following sections describe the fate and transport of chemicals in and between media.

5.4.1 Groundwater Transport

Groundwater generally flows in a east to west direction from the uplands east of the WRL Site to the Killbuck Creek floodplain to the west. Groundwater flows beneath Killbuck Creek, but groundwater also discharges to the Creek. The water table is located in a fractured dolomite aquifer beneath approximately the eastern one-quarter of the WRL Site. The main route in which groundwater flows in this aquifer is through fractures in the dolomite. Further to the west (i.e., across the remainder of the WRL Site), the water table is located in the shallow sand and gravel aquifer which overlays the fractured dolomite aquifer. The two aquifers are generally hydrologically connected with each other. Measured hydraulic conductivities ranged from a geometric mean of 7×10^{-3} cm/sec for the unconsolidated aquifer to 2.3×10^{-4} cm/sec for the dolomite aquifer. Based on the VOC distribution observed in the groundwater monitoring wells and the measured permeabilities of the aquifer soils, groundwater movement is expected to be the primary migration pathway for the contaminants detected at the WRL Site.

Leachate from the landfill represents a potential continuous source of groundwater impact, which would be expected to decrease over time as the final cover is placed reducing precipitation infiltration into the landfill and as leachate continues to be removed. Transport of VOCs in the unsaturated zone of sand and gravel beneath the landfill or in the sand and gravel aquifer is not expected to lead to significant adsorption to these materials. Attentuation of metals and SVOCs is expected to be greater than that of VOCs, because of their lower mobility. Attenuation due to dilution can be expected to reduce concentrations of contaminants in groundwater in the downgradient direction.

The following is a discussion by chemical classification of the fate and transport mechanisms for the primary contaminants present in the groundwater underneath the WRL Site. A listing of individual chemicals within each classification, as well as a discussion of their relevant fate and transport parameters, is presented in Sections 5.3.1 and 5.3.2.

Chlorinated Ethenes and Ethanes - Chlorinated ethenes and ethanes were found in groundwater both up and downgradient of the WRL Site. Because of their solubilities in water and relatively low soil adsorption coefficients, these compounds would not be effectively attenuated and should continue to migrate from east to west at a velocity approximate to groundwater. Chlorinated ethenes and ethanes have been detected at monitoring well G116A west of Killbuck Creek. Anaerobic conditions expected to be present in the groundwater would result in an increase in cis- and trans-1,2-dichloroethene and vinyl chloride concentrations with time, which would in turn result in a decrease in concentrations of the parent compounds trichloroethene, tetrachloroethene, and 1,1,1-trichloroethane. Cis- and trans-1,2-Dichloroethene also degrade, but more slowly than trichloroethene and tetrachloroethene. Vinyl chloride has a long biodegradation half-life and should persist over a long period of time, but will eventually degrade under aerobic conditions.

BETX - BETX compounds were found both up and downgradient of the WRL Site, but at lower concentrations than the chlorinated compounds. Benzene is the most mobile of these compounds. Toluene and xylene can also migrate in groundwater, but should be attenuated to a greater degree than benzene and the chlorinated compounds. BETX compounds should continue to migrate from east to west, but at a velocity less than groundwater. BETX compounds were not detected in monitoring well G116A west of Killbuck Creek. Aerobic and anaerobic degradation pathways are both possible for BETX compounds. Anaerobic degradation intermediates include derivatives of cyclohexane and aliphatic hydrocarbons (Dragun, 1988). Hydrolysis of aerobic degradation products may yield various benzene derivatives.

Chlorinated Benzenes - 1,4- and 1,2-dichlorobenzene were found both up and downgradient of the WRL Site. The chlorinated benzenes would be expected to attenuate in soils based on their soil adsorption coefficients. Low levels of chlorinated benzenes may continue to migrate from east to west due to their limited solubility in water and the likely low organic material content of the soils and bedrock. Chlorinated benzenes were not detected in monitoring well G116A west of Killbuck Creek. There is no evidence that biodegradation of these compounds is occurring.

Phthalates/PAHs - Phthalate and PAH compounds had a low frequency of detection in groundwater wells. Based on their soil adsorption coefficients, these compounds should be effectively attenuated and have limited mobility.

Nickel, Arsenic, Barium - Nickel, arsenic, and barium have been found both up and downgradient of the WRL Site. These compounds were found at higher levels in areas corresponding to the chloride plume and appear to be migrating from east to west with groundwater. Barium does not readily form precipitates and is typically mobile. Nickel is often highly mobile due to its ability to complex with soil organic materials which are soluble in water. Arsenite is expected to be the predominant form of arsenic since the likely anaerobic conditions of the landfill would result in a low oxidation/reduction potential soil environment. Arsenite is the more mobile and toxic form of arsenic. Migrating metals will either be attenuated, or discharged to the waters or sediments of Killbuck Creek.

Lead, Cadmium - Both lead and cadmium were found in three wells around the perimeter of the landfill. These compounds do not appear to be migrating at significant concentrations from the landfill. Both lead and cadmium are likely to be attenuated through fixation or adsorption mechanisms within the soil matrix. Migration of lead and cadmium should remain limited to the immediate boundaries of the landfill.

PCBs, Pesticides - Low levels of PCBs and chlorinated pesticides were detected in the landfill leachate, but were not detected in any groundwater samples. It appears that these compounds have been effectively attenuated to below analytical detection limits prior to reaching groundwater.

In general, assuming the landfill is closed as currently scheduled, with a cap in place and an upgraded leachate and gas collection systems operating, transport of contaminants to the groundwater would be expected to reduce because infiltration would be significantly reduced and leachate would continue to be extracted. Potential degradation of the liner over time should have limited impact under these conditions.

5.4.2 Transport of Contaminants From Soil Vapor Phase

The VOCs discussed in Section 5.3.1 would be expected to exist in both the dissolved and vapor phases. SVOCs discussed in Section 5.3.2 are also typically present, to a lesser extent, in the soil vapor phase. Potential pathways of migration for the soil vapor include emission to the atmosphere via vertical migration out of the landfill, adsorption onto the soil phase or dissolution into groundwater, and removal by the gas extraction system.

Equilibrium between the soil vapor and leachate or groundwater phases is expected to follow a Henry's law relationship for dilute solutions. Contaminant transport via dust is also discussed in this subsection.

For reasons discussed in more detail in Section 4.7.2, vapor migration resulting from equilibrium with organics present in the landfill leachate, although it cannot be completely ruled out, is not expected to be a primary transport pathway for groundwater VOC contamination based on present and past landfill conditions and leachate composition. A discussion of organic compounds which exist in the vapor phase is presented in Sections 5.3.1 and 5.3.2.

Landfill gas extraction is expected to be the primary vapor migration pathway. Subsurface landfill gas migration has previously occurred, but a landfill gas collection system has been operational since 1980. In December 1984, the system was expanded to 70 extraction wells with the gas burned as a fuel source for the sludge drying plant. The system currently consists of 91 extraction wells controlling landfill gas migration. As discussed in Section 4.6, emissions to the atmosphere resulting from migration vertically through the landfill do not appear to be a primary pathway. Low level emissions will be quickly attenuated during downwind transport.

In the future, after the closure of the WRL Site, it is anticipated that the gas collection system will continue to be maintained until gas generation is substantially reduced. For this reason, releases of gas to air would not be expected to be a substantial route of transport for volatile contaminants from the landfill under current and probable future conditions.

During normal operations of the landfill, waste is processed and covered with clean fill on a daily basis. Substantial amounts of fugitive dust may be generated at a landfill as the large trucks and machinery drive on the access roads and process the waste. In the case of the WRL Site, the access roads have been covered with bituminous materials to minimize dust generation. Within the fill area, water is sprayed on the surface of the work area when necessary (i.e., dry conditions) to reduce the release of fugitive dusts. As sections of the landfill reach their final elevation, a vegetated cover will be constructed. As suggested by Cowherd (1985), if a surface is vegetated year round, dust generation will be substantially reduced. Because dust minimization technologies are used to reduce dust generation during waste processing, and a clean soil cover is placed over the processed waste, dust generation and transport by air would not be expected to be a substantial means of contaminant transport at the WRL Site under current conditions.

5.4.3 Transport of Contaminants in Groundwater to Killbuck Creek

Downgradient of the WRL Site, the shallow unconsolidated aquifer is hydraulically connected with Killbuck Creek. Based on staff gauge readings in Killbuck Creek and water table measurements of wells near the Creek, groundwater is discharged to the Creek. Groundwater also passes underneath the Creek and continues in a western direction. Based on measurements of chemical concentrations in Creek surface water and sediment, the WRL Site has not impacted the Creek.

If shallow contaminated groundwater were discharged to the Creek, it is expected that significant dilution would occur. Volatile organic compounds would be volatilized to the air. Dissolved metals and SVOCs may be absorbed onto suspended sediment and deposited downstream. Biodegradation of organic compounds may also occur. If the Killbuck Creek sediments were contaminated, they may be transported downstream to some extent, mostly during floods. Killbuck Creek passes through the Killbuck Creek Forest Preserve and then merges with the Kishwaukee River about 2 miles downstream.

5.4.4 Transport of Contaminants in Surface Soil or Waste to Killbuck Creek

Surface soil and waste samples were not collected during the RI. Therefore, no data is available to quantitatively assess the transport of waste from the fill area via surface erosion. Qualitatively, transport of contaminants via surface erosion caused by rainfall would be expected to be minimal under current land use conditions. The newly processed waste is covered by the end of the day with clean fill; therefore, only a small area of waste would be susceptible to erosion. Older areas of waste are covered with clean fill, so only clean soil would be available for surface erosion. Under current conditions, erosion of surface soil by storm water runoff would not be anticipated to be a substantial means of contaminant transport.

A clay cap with vegetative cover will be installed as sections of the landfill are closed. For this reason, the transport of contaminants via surface soil erosion would not be expected to be a substantial means of contaminant transport at the WRL Site.

SECTION 6
BASELINE RISK ASSESSMENT

6.1 Introduction

The Baseline Risk Assessment (BRA) characterizes the potential threat to human health and the environment posed by Site contaminants, assuming no remedial action is taken at the Site (i.e., the "No Action" alternative). This assessment provides information which will assist in evaluating possible remedial measures for the Site, by providing a baseline estimation of potential health risks. Preparation of the BRA utilized the detailed guidance on conducting risk assessments provided in the Risk Assessment Guidance for Superfund (RAGS) (U.S. EPA, March 1989 and December 1989).

Two health risk scenarios were evaluated for the Site. The first scenario assumes that current land use conditions prevail and that no action is taken to remediate the Site. The second scenario considers reasonable assumptions of future use of the Site if no remedial action and no institutional controls (e.g., deed restrictions) are placed on the Site.

In general, the objective of the BRA, as outlined in the RAGS, is to characterize the extent of contamination and the potentially exposed population(s) sufficiently to determine what risks need to be reduced or eliminated and what exposures need to be prevented. This objective may be attained by identifying and characterizing the following parameters:

- Contaminants present in various environmental media (e.g., air, soil, surface water, and groundwater)
- Contaminant toxicity
- Environmental fate and transport mechanisms, including physical, chemical, and biological properties, that influence the contaminant(s)
- Contaminant migration routes and potential exposure pathways (to receptors)
- Potential receptors (e.g., human)
- Potential and extent for receptors to be exposed
- Comparison of exposures to acceptable levels based on available regulatory and toxicological information

These parameters are organized into the following sections within the BRA:

- Chemicals of Potential Concern
- Exposure Assessment
- Toxicity Assessment
- Risk Characterization

The risk assessment process progresses from a selection of chemicals that pose the greatest potential health threat and/or are representative of the compounds being released from the Site, through determination of populations potentially exposed to releases, to identification of the potential toxicological effects to exposed populations, and finally, to the estimation of corresponding risk to the exposed populations.

6.1.1 Selection of Chemicals of Potential Concern

Chemicals of Potential Concern were selected based on their presence in media at the WRL Site (west of Lindenwood Road) without regard to source(s) of those chemicals. The source(s) of chemicals present in the groundwater in the southeastern corner of the Site will be evaluated by additional studies. A chemical was retained as one of potential concern even if U.S. EPA critical toxicity values were not available. In those cases where critical toxicity values were not available, a qualitative assessment of the health risk associated with the chemical was made.

6.1.2 Exposure Assessment

The exposure evaluation includes identification of actual or potential routes of exposure, characterization of the exposed populations (receptors), and a determination of the extent of exposure by estimating contaminant intakes.

Populations which may potentially be exposed to Site contaminants through the various routes of exposure (i.e., direct contact, inhalation, ingestion) include human populations and sensitive subsets of human populations (e.g., children, elderly). Because human health risks are determined on a lifetime basis, various stages in a person's life are modeled. In this assessment, a 70-year lifetime is assumed to be comprised of 15 years as a child and 55 years as an adult.

To quantitatively assess health effects associated with releases from the Site, the amount of contact with chemicals of potential concern by the receptors was determined. Human exposure was expressed in terms of a chronic daily intake (the amount of substance contacted, inhaled, or ingested per unit body weight each day). Where appropriate, chronic daily intakes were calculated for multiple routes of exposure to a chemical or only for specific routes of exposure, if route-specific toxic effects were noted.

6.1.3 Assessment of Toxicity

This section identifies known and inferred adverse health effects associated with exposure to those chemicals of potential concern that were calculated to contribute substantially (i.e., >5%) to the total risk for a given medium at the WRL Site. Toxicological information from the scientific literature, including epidemiological studies, animal studies, mutagenicity assays, and structure-activity relationship studies, were reviewed and interpreted to provide a summary of potential health effects. The relationship between the level of chemical exposure and the magnitude of the toxic effect (dose-response relationship) for each chemical is considered by utilizing critical toxicity values (e.g., reference doses and carcinogenic slope factors) developed by the U.S. EPA. These toxicity values are derived from the most appropriate toxicological information obtained from animal studies or human studies, when available. Uncertainty factors are incorporated in these toxicity values to account for numerous uncertainties involved in their derivation (e.g., extrapolation from studies on animals to humans).

6.1.4 Risk Characterization

The risk characterization process integrates findings from the exposure assessment and toxicity assessment sections of this report. Estimates of excess cancer risk for carcinogenic chemicals were made utilizing cancer slope factors (SF) developed by U.S. EPA. Risk to the noncarcinogenic effects of chemical exposure are estimated by comparison to reference doses (RFD) developed by U.S. EPA.

6.1.5 Background

The previous chapters of the RI Report provide descriptions of the Site location, history, and physical characteristics (i.e., geology, hydrogeology, etc.) and an evaluation of the chemical constituents found in the following media:

- Air
- Sediment
- Surface Water
- Groundwater

Appropriate sections of the RI Report should be consulted for detailed descriptions of sample locations and the chemical and physical properties of the various media.

6.2 Chemicals of Potential Concern

More than 50 organic and inorganic chemicals from U.S. EPA's Target Compound List (TCL) and Target Analyte List (TAL) were detected in samples taken during the RI. In some samples, tentatively identified compounds (TICs) were also detected. The minimum and maximum contaminant concentrations and frequency of detection of the TCL chemicals are presented by environmental medium in Table 6-1. Groundwater indicators (e.g., total phenolics, chlorides), with the exception of nitrate + nitrite nitrogen, were not included in Table 6-1. These parameters were used as indicators of groundwater contamination, but cannot be used as parameters to assess the risk to groundwater. The detailed analytical results for each chemical listed in Table 6-1 are presented in Appendix C.

6.2.1 Data Validity and Use

Due largely to the difficulty in the analysis of chemicals in environmental media, the usability of the analytical results varied. A discussion of the quality of the analytical data and the process used to assess usability is presented in Appendix H.

Most frequently, data were determined to be estimated and were qualified with "J". The degree of the uncertainty and the direction of the bias (under or over estimation) associated with the data are not the same for each estimated value and are often difficult to ascertain for individual samples. Table 6-2 lists examples of conditions in which organic compound analytical results would be qualified as estimated. In the BRA, estimated data were used with unqualified data (data with no qualifier) for the WRL Site analysis except when data was considered of such poor quality that they should not be used (i.e., ambient air data). Generally, inclusion of estimated values would result in a more conservative (higher) estimation of risks than if samples with estimated values were considered to contain no detectable contaminants and were not used.

As outlined in Appendix H, the quality of the ambient air data is limited by a number of deficiencies associated with the analysis of these samples. These include holding time exceedences and trip blank and method blank contamination. In addition, field duplicates, while collected at different flow rates, had similar chemical masses. This would indicate the chemicals may have been introduced on the adsorptive medium after sample collection. Due to the limited confidence in the data quality, and based on professional judgement, the ambient air data were not considered of sufficient quality for use in the BRA to quantitated risks. Rather, the data are used as a qualitative indicator of the level of chemical release to ambient air from the WRL and exposure to residents. This is consistent with Guidance for Data Usability in Risk Assessment (U.S. EPA 1990).

6.2.2 Site Definition

The WRL Site is situated immediately adjacent to another Superfund site (Acme Solvent), and contamination has been detected at both Sites. The objective of this BRA is to assess risks at the WRL Site regardless of the source(s) of contamination. Therefore, for evaluation purposes, sampling locations west of Lindenwood Road are considered to represent the WRL Site.

The following wells were considered on-Site or downgradient of the Site, and off-site, respectively.

- Wells on-Site or downgradient of the WRL (evaluated as WRL Site locations) - Wells G107, G114, G110, B13, P6, P7, B14, G115, P4R, P3R, P1, MW106, G116A, G116, G117, G119, G119A, G118R, G118A, B15, B15P, B15R, B12, G109, G109A, G111, G113, G113A, and PW1.
- Off-Site Upgradient Wells - All wells east of Lindenwood Road (G112).

During the RI, upstream and downstream samples were taken in Killbuck Creek. Surface water and sediment sampling locations SW01-SW04 and SD01-SD04 were located downstream of the WRL Site, and were evaluated as WRL Site locations. Sample location SW05/SD05 was located upstream of the WRL Site and was considered to represent background.

6.2.3 Procedures for Selecting Chemicals of Potential Concern

Chemicals of potential concern were selected based on the following five criteria, as outlined in RAGS:

- Positively detected in at least one Contract Laboratory Program (CLP) sample in a given medium
- Detected at levels significantly elevated (i.e., factor of 2) above levels of the same chemicals detected in associated blank samples
- Detected at levels elevated above naturally occurring levels of the same chemicals
- Only tentatively identified, but either may be associated with the Site based on historical information or have been confirmed by SAS
- Transformation products of chemicals demonstrated to be present

Those chemicals that met one of the five initial selection criteria were considered chemicals of potential concern. The exceptions to these rules were for those chemicals detected in landfill leachate, but not in other media at the WRL Site, and chemicals for which U.S. EPA critical toxicity values have not been developed and were unavailable from the Environmental Criteria and Assessment Office (ECAO) at the time this report was prepared. Explanations for these exceptions are provided in Section 6.2.5.

Landfill leachate production and handling is a routine process associated with municipal landfills and is not unique to the WRL Site. The BRA focuses on those issues which are germaine to assessing actual releases of contaminants which are not associated with typical landfill operations (e.g., groundwater contamination). Leachate chemical data are used to determine a fingerprint of those contaminants which may be released to media, such as groundwater, which the general public may become exposed to off-site due to chemical transport. The BRA assesses risks to actual releases; therefore, if a chemical is found in leachate, but not in potentially affected media, it is assumed a release has not occurred for purposes of the BRA. A chemical is not selected as a chemical of potential concern solely because it was detected in leachate.

For those chemicals considered to be of potential concern at the Site, but for which there were no critical toxicity values, a qualitative assessment was made of their toxicity. Chemical concentrations in media were compared to health based criteria (e.g., MCLs), where available, to make a semi-quantitative assessment of health risk for those chemicals which, based on their toxicity, may pose a health risk.

6.2.3.1 Chemicals of Potential Concern. The process described in Section 6.2.3 was used for selection of the chemicals of potential concern (see Table 6-3). Table 6-3 indicates the media in which each chemical was detected. For each medium in which the chemical was detected, with the exception of leachate, a rationale is given for either including or excluding the chemical from the final list of chemicals of potential concern. Leachate data were considered qualitatively in the selection of chemicals of potential concern, but as mentioned previously, a chemical was not selected as a chemical of potential concern solely because it was detected in leachate.

CHEMICALS OF POTENTIAL CONCERN

Volatile Organic Compounds	Metals/Inorganics	Semi-Volatile Compounds
Acetone	Arsenic	Bis(2-ethylhexyl)phthalate
Benzene	Barium	Di-n-butylphthalate
Bromoform	Cadmium	1,2-Dichlorobenzene
Bromodichloromethane	Chromium	1,3-Dichlorobenzene
Carbon Tetrachloride	Cobalt	1,4-Dichlorobenzene
Chlorobenzene	Copper	Diethyphthalate
Chloroethane	Iron	PAHs(noncarcinogenic)
Chloromethane	Lead	
Chloroform	Manganese	Pesticide/PCBs
Dibromochloromethane	Nickel	None
1,1-Dichloroethane	Nitrate + Nitrite	
1,2-Dichloroethane	Silver	
1,1-Dichloroethene	Sodium	
1,2-Dichloroethene (cis&trans)	Thallium	
1,2-Dichloropropane	Vanadium	
1,3-Dichloropropene	Zinc	
Ethylbenzene	Cyanide	
Methylene Chloride		
Tetrachloroethene		
1,1,2,2-Tetrachloroethane		
Toluene		
1,1,1-Trichloroethane		
Trichloroethene		
Vinyl Chloride		
Xylenes (o-, m-, and p-)		

6.2.3.2 Chemicals of Potential Concern at the WRL Site Associated with Anthropogenic Background. Some chemicals of potential concern at the WRL Site are potentially associated with upgradient/upstream anthropogenic background. Many chemicals detected in groundwater at the WRL Site have been detected in Site leachate and off-Site wells upgradient of the WRL. Therefore, a clear distinction of the source of the chemical contamination cannot be determined. Table 6-4 lists those chemicals that were detected in the WRL leachate and groundwater at the WRL Site and off-Site upgradient wells.

As with groundwater, a number of chemicals detected in Killbuck Creek surface water and sediment may be associated with other upstream sources, as well as the WRL Site. Table 6-5 lists the chemicals that may be associated with both the WRL and other sources.

6.2.4 Tentatively Identified Compounds

Tentatively identified compounds (TICs) were considered qualitatively in the BRA, but because of the uncertainty inherent in TIC identification, quantitation, and general lack of toxicity information on TICs, little weight was given to these compounds in the BRA. Tables 6-6, 6-7, and 6-8 summarize the TICs detected in leachate, groundwater, and sediment (respectively), the frequency of detection, and the minimum, maximum, and average concentrations within each medium. The following is a summary of the distribution of TICs in each medium at the Site.

6.2.4.1 Leachate. The TICs detected in WRL Site leachate (see Table 6-6) may be associated with the degradation of municipal waste and organic sludges that the landfill accepts as waste. The majority of the TICs detected were semi-volatile compounds normally associated with the degradation of organic matter (e.g., total phenolics, carboxylic acids).

6.2.4.2 Groundwater. Thirty-six TICs were detected in groundwater in monitoring wells near the WRL. Most TICs were detected only once, but fourteen TICs were detected more frequently (e.g., ethyl ether) (see Table 6-7). Most TICs in leachate (i.e., potential source of groundwater contamination) were not consistently detected in each of four rounds of groundwater sampling. The TICs that were detected in groundwater were found at low concentrations (i.e., generally 5 to 30 ug/L). As in leachate, the majority of TICs detected in groundwater were semi-volatile compounds potentially associated with organic matter degradation.

6.2.4.3 Sediment. Five TICs were detected in Killbuck Creek sediment (see Table 6-8). Of the TICs detected in sediment, none had been found in groundwater. The downstream TIC detections in the potentially affected part of the Creek occurred at a single location (i.e., SD02). TICs were also identified in the Creek at the upstream background sampling station (i.e., SD05). The TICs upstream were different than the TICs detected downstream.

6.2.4.4 Surface Water. No TICs were detected in surface water of Killbuck Creek.

6.2.4.5 Summary. Based on the distribution of TICs in media, it appears that fourteen TICs associated with leachate, as a result of waste degradation, have entered the groundwater. In general, these TICs are semi-volatiles having a low to moderate toxicity to humans. Although TICs were detected in groundwater and sediments, none of the TICs were similar between the two media. No TICs were detected in Killbuck Creek surface water. It appears from the data that TICs have not been released into Killbuck Creek from upgradient sources of groundwater contamination.

Consistent with RAGS, these TICs were not selected as chemicals of potential concern, because they could not be clearly linked to the Site based on historical information. In addition, there is generally a lack of toxicity information for TICs and a degree of uncertainty in identification and quantification of these compounds. As part of the risk assessment process, ECAO was contacted to determine if toxicity values were available for the TICs at the WRL Site, but no information was made available.

6.2.5 Chemicals Not Selected as Contaminants of Potential Concern

The health risks associated with exposure to a number of chemicals could not be assessed quantitatively due to the lack of appropriate toxicology information. The exclusion of these chemicals, on a quantitative basis, is not expected to substantially alter the outcome of this BRA for one or more of the following reasons:

- The chemicals were detected at low concentrations
- The chemicals were detected in a small number of samples
- The chemicals have a low order of toxicity
- Physical/chemical properties (Table 6-9) suggest some chemicals are not environmentally mobile

The health risks associated with chemicals of potential concern that could not be addressed quantitatively in the BRA will be addressed qualitatively in Section 6.5.3.3.

6.3 Exposure Assessment

The aim of the exposure evaluation is to arrive at an estimation of the magnitude of contaminant intake by exposed populations. This estimation integrates information on pathways of contaminant migration within the environment (see Section 5), concentrations of contaminants at points of contact with receptors (see Table 6-14), estimates of the degree of receptor contact with the contaminated media (see Sections 6.3.1.2 and 6.3.2.2), and factors which address the efficiency of contaminant entry into the body (e.g., bioavailability factors) (see Tables 6-12 and 6-18). This assessment is performed using assumptions of population activities based on current land use conditions at the WRL Site and predictions of possible future conditions at the WRL Site.

In this section of the risk assessment, potential pathways by which populations may be exposed to Site contamination are evaluated. For a chemical to elicit an effect, an individual must be exposed to the chemical via ingestion, inhalation, or dermal contact. The concentration, duration, and frequency of chemical exposure will determine whether an effect in the individual may occur. Therefore, it is necessary to estimate the exposure conditions at the WRL Site.

The results of Site visits conducted during the RI were used to develop estimates of exposure under current Site conditions (i.e., human activity patterns). Current patterns of land use and zoning and historical patterns of land use were used as a basis to estimate the potential future land use of the contaminated areas.

Several potentially exposed populations (e.g., children, adults, elderly) were considered, but exposure estimates were based on the most sensitive subgroup to simplify the exposure assessment. An equivalent dose of a chemical is more toxic to children than it is to adults; therefore, childhood exposure estimates of chemical intake were calculated for a medium, unless adults would be the primary exposed population. An exception to this is exposure to chemicals via drinking water. In this case, persons may be exposed to the water over a substantial portion of their lifetime (i.e., both as an adult and as a child). Therefore, the amount of exposure through drinking water is calculated using lifetime average rates of water ingestion and body mass. In this way, the average chemical exposure during childhood and adulthood is estimated (time-weighted) for the exposure period.

The following subsections describe the exposure potential at the Site based on current and probable future Site conditions (also see Table 6-13).

6.3.1 Potential Exposure Pathways Based on Current Land Use Conditions

The following is a description of the current land use at the WRL Site and surrounding area, as well as the chemical exposure potential from specific media.

6.3.1.1 Description of Current Land Use Conditions. The WRL Site is located at the southern edge of Winnebago County, approximately 5 miles south of the City of Rockford. The area has a climate typical of the midwestern United States. The spring, summer, and fall months are the warmest with temperatures averaging 71°F during summer. In the winter, the average air temperature is 23° F. The area receives an average of 38 in. of precipitation per year, and an average of 33 in. of snowfall.

The area surrounding the landfill is characterized by subdued rolling hills rising above alluvial valleys. The landfill is situated on a 60-acre parcel on a topographic high between Killbuck Creek to the west and unnamed intermittent streams to the north and south. Lindenwood Road borders the landfill property to the east.

Killbuck Creek, a perennial stream, flows within 250 ft of the western WRL Site boundary and merges with the Kishwaukee River about 2.5 miles to the north. In the area of the landfill, the Creek is approximately 50 ft wide, 1 ft to 3 ft deep, and flows to the northwest. The confluence of Killbuck Creek and the northern intermittent stream is about 1000 ft northwest of the Site, and the confluence of the southern intermittent stream is about 1200 ft south of the WRL Site.

Killbuck Creek is used as a recreational area approximately 1 1/2 miles downstream of the landfill (Killbuck Bluffs Forest Preserve). The Creek's 100-year floodway varies greatly in width (approximately 30 to 750 ft) on both the eastern and western sides of the channel due to the meandering path of the Creek bed near the landfill. The shortest downgradient distance from the landfill to the western edge of the Creek's floodplain is approximately 1000 ft.

Based on a current wetlands inventory (ENCAP 1990), a 3.73-acre area of wetlands exists on-Site. The wetlands are located approximately 150 feet south of the landfill, and extend to the west into the creek's floodplain. The wetlands are composed of 3.18 acres of scrub-shrub/forested wetland, 0.46 acre of shallow drainage way, and 0.09 acre of farmed wetland.

The unconfined (shallow) aquifer in the area of the WRL Site is used as a water resource by private residences near the WRL Site. The closest private well is PW1 (also identified as PWO), located approximately 0.4 mile southwest of the landfill. Private wells in the area west of the landfill for which well construction reports were available generally drew water from the unconsolidated water table aquifer (i.e., sand and gravel).

The land use around the WRL Site is a mixture of industrial, agricultural, commercial, and rural residential. The predominant use near the Site is agricultural to the north, south, and west. East of the Site is the former Acme Solvent facility, which disposed of wastes generated at the Acme Solvent reprocessing plant. The Acme Solvent Site is listed on the National Priorities List. A ROD was issued in December 1990. VOCs are present hydraulically upgradient of the WRL Site and are affecting the background water quality beneath the WRL Site.

An inactive alcohol production plant (all equipment removed) and an active sewage sludge drying plant are located north of the WRL. The Rockford Skeet Club is located across Lindenwood Road to the northeast. There are residences north, south, southwest, and southeast of the Site. The residents in the area generally use their own private wells for drinking water. Five private wells located hydraulically upgradient of the WRL Site have been chemically contaminated. Some of these residents have had treatment systems installed, as part of a consent order to some of the Acme Solvent PRPs, to remove the chemical contamination from the water (Harding-Lawson Associates, 1990).

Census information (i.e., census tract data) for the area surrounding the WRL Site was used to characterize the potential human receptors by age. Based on census tract information for Valley Township in Winnebago County collected during the 1980 census, there are many families (i.e., potential receptors) with children living near the landfill. Based on 1980 data, approximately 888 families live in the census tract that contains the landfill. Children between the ages of 0 and 15 comprised 31 percent of the total population of 3575 persons within the tract. Other tracts which surround the Site had a similar demographic make-up in 1980, although in general, the areas farther north (City of Rockford) had larger population bases.

The WRL Site is an operating municipal landfill. A gas collection system to limit methane gas migration off-site, is in place in areas where refuse has already been placed. The gas is used as a fuel source for the sludge driers which are located north of the active fill area. Leachate produced by the landfill is stored in an on-Site lagoon and aerated to facilitate biological degradation of the organic components. Leachate is pumped periodically from the lagoon and shipped off-Site for treatment.

Access to the WRL Site is currently restricted. Chainlink fencing and a secured gate are used to limit access to the landfill and limit unauthorized dumping. The fence does not present a continuous barrier surrounding the entire landfill, and therefore, people have the potential to trespass on-Site. Currently, the level of human activity at the Site is minimal, with the exception of the facility workers and customers who dispose of waste at the facility. If nearby residents trespassed on-Site, the exposure potential to waste and leachate would be typical of other active municipal landfills and not unique to the WRL Site.

6.3.1.2 Media Specific Exposure Potential - Current Land Use Conditions. The following sections describe the potential for chemical exposure to humans from air, water, sediment, waste, leachate, and food near the WRL Site under current land use conditions.

Air. Currently, the WRL Site has a landfill gas collection system. The gas is used as a fuel source to dry sludge on-Site. As the fuel is burned, the volatile organic chemicals in the gas stream are destroyed. Burning the gas effectively eliminates substantial fugitive release of chemicals to air.

The lagoon which collects leachate at the WRL Site is aerated to facilitate microbial degradation of the organic fraction of the leachate. During aeration, there is the potential for release of chemicals to the atmosphere. Although chemicals may be released from the lagoon, dispersion and dilution of the chemicals in air would minimize their concentration downwind of the WRL Site.

Based on ambient air data collected during the RI, a quantitative determination of whether the WRL is affecting ambient air cannot be made. Due to the limited confidence in the data (see Section 6.2.1), the ambient air data are used qualitatively in the BRA. The data are used to assess whether the level of chemical exposure from ambient air to nearby residents appears substantial and warrants inclusion as an important exposure pathway in the BRA.

VOC concentrations upwind and downwind of the WRL Site were very low, which suggests a low level of VOC exposure to nearby residents. A comparison of the ambient air concentrations to relevant Site exposure levels for workers was made to qualitatively assess the significance of the VOC concentrations detected in ambient air (see Table 4-7). Based on this comparison, concentrations of VOCs in ambient air were many orders of magnitude lower than the safe exposure levels for workers. Thus, the release of chemicals to air via volatilization was not considered a substantial route of chemical exposure to humans under current land use conditions.

The release of fugitive dust due to vehicular traffic is another transport mechanism by which persons downwind of the WRL Site may become exposed to contaminants. Surface soil and waste samples were not collected and analyzed for chemical contamination as part of the RI. Under current Illinois EPA regulations, landfills must control the generation of fugitive dusts using appropriate methods (e.g., wetting of cover material). The WRL controls dust at the landfill, when appropriate (i.e., dry conditions), to minimize fugitive dust releases. For this reason, fugitive dust was not considered a route of chemical exposure to humans under current land use conditions.

Groundwater. The area surrounding the landfill is semi-rural and many private homes in the area use the aquifers in the area of the landfill as a drinking water source. The two predominant aquifers downgradient of the Site are a shallow unconsolidated sand and gravel aquifer overlying a variably fractured bedrock (dolomite) aquifer. The nearest private well (PW1) is located downgradient of the WRL Site and west of Killbuck Creek. Based on testing of water from PW1, the well has not been affected by the WRL Site.

The plume of contaminated groundwater has migrated west of Killbuck Creek and approximately 900 ft downgradient from the western edge of the landfill. Within this area, the water is not used as a drinking water source and there are no drinking water wells within approximately 2,000 ft downgradient of the WRL Site.

Currently, private drinking water supplies downgradient of the landfill have not been affected. Therefore, groundwater is not considered a source of chemical exposure to humans under current Site conditions.

Surface Water and Sediment. In general, Killbuck Creek does not appear to have been affected by groundwater contamination. The chloride plume associated with the WRL Site has migrated west of the Creek, but the amount of groundwater discharge to the Creek appears to be small. Therefore, if contaminated groundwater or surface water had been discharged to the Creek, the resultant chemical concentration in surface water would be very low due to dilution. It appears, from the results of sediment sample SD03 there may have been discharge of VOC-contaminated groundwater to the creek.

Conversations with personnel in the Division of Public Water Supply of the Illinois EPA indicate that Killbuck Creek and the downstream Kishwaukee River are not listed as public water supplies in Winnebago County. Therefore, ingestion of surface water by residents was not considered a source of chemical exposure under current Site conditions.

The section of Killbuck Creek a short distance downstream of the landfill is a designated recreational area. For this reason, it is assumed that persons may use the section of the Creek near the landfill for recreational purposes.

Fish consumption from Killbuck Creek is possible, but the chemicals detected in the Creek (e.g., VOCs) do not generally bioconcentrate in fish. Fish are not expected to become contaminated due to the low levels of VOCs detected in surface water of Killbuck Creek. Therefore, fish consumption was not considered a substantial pathway for chemical exposure to humans.

Killbuck Creek is shallow in depth (i.e., 1 to 2 ft deep), therefore it was assumed children do not swim in the Creek. For this reason, incidental ingestion of Creek water was not considered a substantial source of chemical exposure to children.

Because children (i.e., most sensitive subpopulation) likely wade, splash, and play in Killbuck Creek, they may be exposed to sediment through incidental ingestion and dermal contact. Although there is the potential for incidental ingestion and dermal contact of sediment while wading, most sediment is likely to wash off children while they are wading. The potential for sediment exposure was considered low, however, exposure conditions were estimated for a child (average age 9.5 years).

If children wade in the Creek, they may absorb chemicals through their skin. Therefore, dermal contact with surface water was considered a possible route of chemical exposure to children. The following exposure conditions were assumed for the children who may be exposed to sediments and surface water in Killbuck Creek:

- Children contact the surface water and sediment mainly with their hands, arms, legs, and feet
- Children ingest a small quantity of sediment while playing
- Children are exposed once each week during the spring, summer and fall (i.e., 8 months per year). During the rest of the year the surface water is uncomfortably cold for contact and sediment is covered with snow or a layer of ice that eliminates exposure to these media.

Based on these current Site exposure assumptions, the chemical risks calculated are conservative (i.e., higher than what is probable).

Food. Garden vegetables or agricultural crops (e.g., corn or soybeans) grown in contaminated floodplain sediment can incorporate trace quantities of some chemicals. The land west of Killbuck Creek is currently used as cropland for livestock feed. Private gardens were not observed in the floodplain. During the RI, no floodplain sediment samples were collected at the WRL Site. Based on the nature of contamination (i.e, groundwater), there is little potential for the floodplain sediments to become contaminated in relation to the groundwater contamination. The majority of contaminants detected in groundwater (i.e., VOCs) do not readily bioaccumulate in plants. For these reasons, ingestion of crops grown in the floodplain of Killbuck Creek are not considered sources of chemical exposure under current Site conditions.

Waste and Leachate. Typical of active municipal landfills, leachate is produced from the degradation of the waste. Due to Site access restrictions, exposure to waste and leachate by the general public is unlikely. The WRL Site employees are the most likely population exposed to the waste and leachate. Worker exposure to waste and leachate is expected to be low, because little direct contact with these media occurs and workers generally use gloves and coveralls to protect themselves from contact with waste and leachate.

Because the general public is not anticipated to be exposed to the waste or leachate, and worker exposure is expected to be minimal, leachate and waste were not considered to pose substantial chemical exposure under current Site conditions. Furthermore, these scenarios do not appear germane to assessing the risks associated with documented environmental releases of chemicals from the WRL Site (e.g., groundwater contamination).

Summary. Exposures to air, groundwater, food, leachate, and waste are not considered sources of Site-related chemical exposure under current land use conditions (see Table 6-13). A gas collection system and dust control measures limit air releases of contaminants, private wells downgradient of the WRL Site are not likely contaminated, and private gardens are not evident near the WRL Site. Also, residents are not anticipated to be exposed to leachate or waste due to access restrictions at the WRL Site.

Although surface water and sediment contact and ingestion are unlikely, there is the potential that children may play in Killbuck Creek and be exposed by these pathways. Therefore, based on the current conditions, it is assumed that children are the most sensitive population potentially exposed to surface water and sediments near the WRL Site.

Killbuck Creek does not appear to be substantially affected by the landfill or other upgradient sources of contamination. For completeness, the risks due to surface water and sediment exposure from contaminants detected in these media were quantified, although their potential magnitude is considered low.

6.3.2 Potential Future Exposure Conditions and Related Exposure Pathways

The following is a description of probable future land use at the WRL Site and surrounding area, as well as a discussion of the chemical exposure potential from specific media at the WRL Site.

6.3.2.1 Description of Probable Future Land Use Conditions. To predict the probable future use of the area surrounding the WRL Site and subsequent levels of chemical exposure, the present zoning patterns, property location, and practical considerations were evaluated consistent with RAGS and the NCP.

Currently, the WRL Site is zoned as a special use (SUP) area. The area around the WRL Site is predominantly zoned as agricultural (AG) with a few exceptions. A small parcel of land directly north of the Site is zoned Industrial Heavy (IH) and a large parcel of land to the northeast of the WRL Site across Lindenwood road is zoned under a Special Use Permit. The following are the use restrictions for the particular zones.

- AG Farming and farm orientated pursuits. Single-family dwellings.
- IH Manufacturing in any form provided the use meets Illinois EPA standards.
- SUP Special Use as designated in the parcel's Special Use Permit

The area downgradient of groundwater flow from the WRL Site is composed of a generally flat, open, parcel of land which is heavily vegetated with mixed species of grasses. The low lying portion of this parcel located near Killbuck Creek is part of the stream's floodplain. Based on practical considerations, such as the potential for flooding and building restrictions in areas with a high water table, it is unlikely that homes will be constructed within the floodplain of the Creek. This assumption is supported by the fact that homes are not currently present in the Creek's floodplain along the section of the Creek that flows near the landfill.

A conversation with Mr. Richard Mohaupt, Director of Public Works for Winnebago County (June 6, 1990), indicated that wells and homes may be built within the floodplain, but only under special conditions. If a home is built in a floodplain, the structure must be built on piers or piles that raise the lowest portion of the structure above the 100-year floodplain. He pointed out that septic systems cannot be placed in the floodplain. This constraint eliminates the possibility of building homes in most floodplains. The property between the landfill and the eastern edge of the Creek is owned by the WRL Site. The nearest location downgradient of the landfill and west of the Creek that is located out of the floodplain of the Creek is approximately 1000 ft from the WRL Site. Therefore, it is unlikely that homes will be built closer than 1000 ft downgradient of the landfill, because of septic system restrictions.

Also, in Winnebago County, the construction of wells in areas of known contamination is highly discouraged. Conversations with Mr. Armour Peterson, Environmental Health Sanitarian for the Winnebago County Health Department (June 6, 1990) indicate that all new well construction Site plans must be approved by the Health Department. One of the criteria used in determining whether a well should be placed in a given location is whether there is known groundwater contamination in the area. The area near the WRL Site was considered to be an area of groundwater contamination concern. Environmental Health personnel can only discourage well construction in areas of known groundwater contamination. If a well was constructed and contamination was detected which would be a health concern, the Health Department has the authority to condemn the well. Therefore, although it may be possible to construct a well near the downgradient side of the landfill, people who choose to do so would face the dual obstacles of the flooding potential and discouragement they would receive from the County Health Department.

In the immediate future, the WRL Site will continue to be used as a municipal landfill. It is estimated that there is approximately 5 to 6 years of capacity remaining. The eventual closure of the landfill in the future will likely include capping and maintenance of the gas and leachate collection systems. Because of the potential for long term differential settling of materials in the landfill, construction of residential structures on the landfill is unlikely. Also, because of the nature of the fill area (i.e., steep side slopes, and presence of gas and leachate collection systems), the landfill is not desirable for residential development. For these reasons, it is unlikely that residences will be built on the fill area in the future. Therefore, a residential Site development scenario was not considered practical for this risk assessment.

6.3.2.2 Media Specific Exposure Potential - Future Land Use Conditions. The following sections describe the potential for chemical exposure to humans from air, water, sediment, and food near the WRL Site in light of the probable future land use conditions.

Air. In the future, it is expected that the gas collection system will continue to effectively minimize releases of VOCs to the air throughout the period in which landfill gas is actively generated. The WRS plans continued use of the landfill gas as a fuel for the on-Site sludge drying process (current contract with the City of Rockford sewage plant to accept sludge through the year 2003). In addition, landfill gas must be monitored for a minimum of 15 years following landfill closure in compliance with Illinois solid waste landfill regulations.

As the landfill ages, the amount of gas production will decrease as organic materials are degraded. Landfill aging will reduce the amount of VOCs that potentially could be released. For the WRL, the period of substantial gas production was predicted to be approximately 20 years post-closure.

If the gas collection system became inoperative (i.e., equipment failure), there is the possibility that VOCs would be released to the atmosphere. The amount of time the gas collection system may be inoperative due to general maintenance or repair is anticipated to be relatively short (days to weeks) and would potentially result in short-term, low concentration VOC exposure to nearby residents.

Following the period of active landfill gas generation, release of VOCs via diffusion through the cover is expected to be substantially less than during the period of greatest gas generation. VOCs released at this time are expected to be diluted appreciably downwind of the WRL Site and are not likely to result in a substantial exposure to nearby residents. Therefore, the volatilization of VOCs from the WRL Site is not considered a substantial source of chemical exposure to humans under future land use conditions.

In the future, the landfill will have a vegetated cover (i.e., after closure), which will minimize dust generation (closure requirement). For this reason, fugitive dust generation is not considered a substantial source of chemical exposure under future land use conditions.

Sediment and Surface Water. As stated earlier, Killbuck Creek is not a source of drinking water; therefore, future exposure to surface water (and sediment) may be limited to recreational use of the stream. Levels of surface water and sediment contamination may remain similar to present levels of contamination. Assuming steady-state conditions will prevail, the risks posed by exposure to these media would be similar to the risks addressed under current Site conditions (refer to Section 6.3.1.2). Although steady-state conditions are assumed for the future, depending upon Site conditions, surface water impacts may be lesser or greater, leading to less or more potential for human exposure.

Groundwater. In the future, it is not probable that existing private drinking water wells will be affected by groundwater contamination associated with the landfill. The distance to the nearest downgradient well is approximately 2,000 ft, and groundwater contamination is currently attenuated within approximately 900 ft downgradient of the Site (i.e., in a westerly direction). Also, because development near the downgradient side of the landfill is difficult due to a high water table and concerns of seasonal flooding, and may be discouraged by public health officials in areas of known groundwater contamination, it is unlikely that wells for drinking water will be constructed directly adjacent to the Site.

It is apparent that private wells are not likely to be affected in the future by the landfill. However, for assessment of baseline conditions, the hypothetical risk associated with the present levels of groundwater contamination was addressed in the BRA, assuming exposure occurs in the future.

In the future, there is the potential for the magnitude and extent of groundwater contamination to change downgradient of the WRL Site. Without remedial actions taken, over the short-term, it is possible that groundwater contamination may extend further west. Over the long-term, it is likely groundwater contamination will be curtailed as the landfill ages and leachate chemical concentrations decrease. Future groundwater impacts due to contamination have not been predicted in this analysis. It is recognized that the hydrogeologic system at and near the Site is complex and dynamic. However, to assess potential future Site risks due to exposure to contaminated groundwater at the Site, it was assumed that groundwater contamination would continue in the future equal in magnitude to the current groundwater contamination at the Site.

In the BRA, it was assumed that people in the future may ingest groundwater with contamination equal in magnitude to what is currently present in downgradient monitoring wells. This assumes that the contaminated groundwater plume moves downgradient to well users in the future at its present concentration (i.e., no further dilution) and/or that wells will be constructed directly downgradient of the Site. Neither of these future circumstances are probable. When interpreting risk estimates, these concentrations are relevant only if exposure occurs at the assumed magnitude in the future. If wells are not constructed near the Site in the future and the groundwater plume does not move further downgradient under prevailing concentrations, to affect existing private wells, groundwater would not be a source of chemical exposure.

The following is a discussion of the assumed exposure conditions for residents who may live downgradient of the Site in the future:

- Residents are exposed to the groundwater for 30 years, which is equivalent to the upperbound estimate (90%) of the time a person lives at a single residence. The area surrounding the WRL Site is semi-rural, with a mixture of farms and residences. Based on area-specific information provided by Mr. Jonathan Holmstrom, most residents near the Site have lived in the area for less than 30 years. Although some persons may live at one residence for longer periods of time, the majority of people live at a given residence for less than 30 years in the United States (EPA, July 1989). Therefore, 30 years was considered a reasonable worst case exposure period for a resident near the WRL Site in terms of groundwater exposure. As suggested in RAGS, the U.S. EPA RPM for the WRL Site was consulted and approved of this exposure period.

- Persons are exposed to the contaminated water via dermal contact, ingestion, and inhalation on a daily basis over the exposure period (i.e., 30 years). Inhalation exposure to groundwater was addressed by utilizing a scientific study which estimates the relative contribution of the inhalation route to the oral route of exposure. According to Cothern, et al. (1986), the inhalation route of exposure may be equally as important as the oral route of exposure in determining risks to groundwater for volatile contaminants. Therefore, to estimate the risk due to air exposure, the oral risk estimate was multiplied by a factor of two for volatile chemicals to account for the contribution of risk associated with breathing VOC contaminated air while bathing. For semivolatiles and inorganic chemicals, volatization was considered to be negligible.

Food. In the future, crops grown on parcels downgradient of the WRL are not expected to become contaminated, because of the nature of contamination present (i.e., groundwater). If contaminated groundwater is used to irrigate crops, there is the low potential that the crops could assimilate trace levels of the contaminants from groundwater. The main group of contaminants likely to be transported in groundwater downgradient beneath croplands are the volatile organic chemicals. During spraying, most of the volatile contaminants would be released to the atmosphere, limiting the amount of contaminant exposure plants would receive. The amount of chemical that would be assimilated by the plant would be expected to be negligible. Human exposure to the crops is unlikely, because most crops grown in the area are probably used for feed for livestock, but if crops were grown for human consumption, the level of chemical exposure would be very low. For these reasons, ingestion of crops grown downgradient of the Site are not expected to present a substantial future pathway of chemical exposure to people.

Waste and Leachate. In the future, the waste will be covered with an engineered cap to eliminate the exposure potential to waste and minimize leachate generation after the WRL Site is closed. In addition, leachate must be managed for a minimum of 15 years following landfill closure as part of post-closure requirements of the recently promulgated Illinois solid waste landfill regulations. Remedial action will be required if groundwater problems attributable to the landfill are indicated. For this reason, exposure to waste and leachate under future use conditions is not considered a pathway of chemical exposure to the general public.

Summary. Under probable future land use conditions, air is not anticipated to be an appreciable pathway of chemical exposure, because the gas collection system will likely remain in place (see Table 6-13). Sediment and surface water exposure is anticipated to be similar to current use considerations (refer to Section 6.3.1.2). Although unlikely, groundwater is considered as a hypothetical source of chemical exposure in the future to residents living downgradient of the WRL Site. It is assumed that residents may be exposed to the groundwater for an extended period of time (i.e., 30 years) on a daily basis. Current levels of groundwater contamination are used as a means of estimating future concentrations of groundwater in private wells.

6.3.3 Exposure Pathways of Greatest Concern at the Site

Risk to human health has been quantitatively estimated for current and hypothetical future conditions at the WRL Site for the following exposure pathways:

Current Site Conditions. Exposure of children to contaminated surface water and sediment in Killbuck Creek.

Hypothetical Future Site Conditions. Exposure of residents living downgradient of the Site to contaminated groundwater.

6.3.4 Estimation of the Extent of Contaminant Exposure

In this section, a quantitative estimate of the chemical intake incurred by a potential receptor is made. Medium-specific concentrations of indicator contaminants at points of receptor contact (exposure point concentrations) are combined with estimates of media intake rates for receptors in each exposure pathway to arrive at the receptor's intake.

6.3.4.1 Exposure Point Concentrations. Values of chemical contamination within each medium were obtained directly from the analytical data.

The Site-wide 95 percent upper-bound confidence limit (95% UBCL) of the arithmetic mean (Gilbert, 1987) or maximum concentration was used to characterize the exposure point concentration for each chemical of potential concern in groundwater, surface water, and sediment. The maximum concentration was used when the 95% UBCL was greater than the maximum value. Using these contaminant concentrations, the calculated risks represent a reasonable worst-case estimate of the exposure point chemical concentrations.

To estimate the exposure point concentrations under current Site conditions, the maximum contaminant concentration in sediment or surface water was used to represent the exposure point concentration to sediment and surface water (95% UBCL > maximum concentration). The data for the four downstream sampling locations (SD01-SD04 and SW01-SW04) were used to calculate the exposure point concentrations of chemicals of potential concern in sediment and surface water.

If a contaminant was not detected at a sample location, the lower of either one half of the contract-required quantitation/detection limit or one half of the lowest reported value detected on-Site was used to estimate the contaminant concentration at that location.

Exposure point concentrations for groundwater under future Site conditions were estimated similarly to surface water and sediment for current Site conditions. To calculate the 95% UBCL in groundwater, data from wells downgradient and sidegradient to the WRL Site were used. Because wells B14 and PW1 did not appear to be affected by the WRL Site or other potential upgradient sources, they were deleted from the database to prevent the artificial dilution of affected wells by these wells. Therefore, the 95% UBCL for groundwater was calculated utilizing data from the remaining 28 on-Site or downgradient wells. The remaining wells were considered off-site and were not used in determining exposure point concentrations.

If a particular well had no detects for a given chemical, the data were handled similarly to the surface water and sediment data. Table 6-14 summarizes the exposure point concentrations of each chemical of potential concern used for calculation of receptor chronic daily intakes due to exposure to groundwater, surface water, and sediment.

6.3.4.2 Estimation of Chemical Intakes. The following sections summarize the equations (obtained from RAGS), assumed media intake rates, and exposure factors used to estimate contaminant intakes. The rates and factors selected are not necessarily representative of all receptors and can be influenced by multiple factors such as lifestyle, age, and gender. Choices made for these parameters were intended to represent conservative estimates of typical or expected intake values. These exposure factors were obtained from the following sources: Exposure Factors Handbook, (U.S. EPA, July 1989) and RAGS. In addition, plausible assumptions about the frequency of contact with sediment and groundwater were made for Killbuck Creek users and future groundwater users based on professional judgment.

The following section describes the calculations used to estimate contaminant intakes for each route of exposure. Intakes of chemicals incurred by individuals are expressed as average chronic daily intakes (CDI).

6.3.4.3 Chronic Daily Intakes. The reference dose (RFD) and slope factor (SF) are values developed by the U.S. EPA, which assume that people will be exposed to the chemical over a chronic exposure period (> 7 years for noncarcinogens) or each day of their lives (70 years for carcinogens). Under most circumstances, the length of exposure to the chemical is likely to be much shorter than chronic or lifetime exposure. Because of this, the CDI likely overestimates the average lifetime intake of the chemical. The CDIs of each chemical of potential concern for each route of exposure to surface water, sediment, and groundwater are summarized in Tables 6-15 through 6-17, respectively.

Based on current U.S. EPA guidance in RAGS, the estimated daily human intake may be adjusted to reflect a chronic or lifetime length of exposure. Chronic daily human intake adjustments differ for carcinogenic and noncarcinogenic chemicals. Chronic daily intakes for carcinogens may be adjusted by multiplying the CDI by the proportion of days in a lifetime that a person is exposed. Chronic daily intakes for noncarcinogens may be adjusted by multiplying the CDI by the proportion of days which a person is exposed over the exposure period. The following subsections present the equations which were obtained from RAGS and were used to estimate CDIs.

Suggested values for many of the exposure variables in each equation (e.g., soil ingestion rate) are described in RAGS and, where appropriate, these factors were applied. These suggested values have been estimated based on a review of pertinent studies by the U.S. EPA. In many instances where suggested values are not available, professional judgment was used in conjunction with Site-specific information to estimate the variable values (e.g., exposure duration).

Incidental Ingestion of Chemicals From Sediment. Direct contact with sediment can result in the incidental ingestion of this material. The CDI of a sediment-bound contaminant ingested by an individual was estimated using the following relationship:

$$CDI = \frac{CS \times IR \times CF \times FI \times EF \times ED}{BM \times AT}$$

Variable		Value Used	Rationale
CDI	= Average Chronic daily intake (mg/kg-day)		
CS	= Chemical Concentration in Sediment (mg/kg)	Lowest value of upper-bound 95% confidence limit (arithmetic mean) or maximum value	- Suggested Value.
IR	= Ingestion Rate (mg sediment/day)	200 (child)	- Suggested Value.
CF	= Conversion Factor (kg/mg)	10^{-6}	
FI[1]	= Fraction Ingested from Contaminated Source (unitless)	1.0	- Assume all sediment ingested from the contaminated area.
EF[1]	= Exposure Frequency (days/year)	35 (child)	- One event per week during 8 non-winter months.

ED[1] = Exposure Duration (years)	10 (child)	- Occurs from ages 5 to 15.
BM = Body Mass (kg)	30 (child)	- 50th Percentile body mass for child (9 to 10 years old).
AT = Averaging Time (days)	Noncarcinogens-3650 (child), Carcinogens -25,550	- Exposure duration for noncarcinogenic effects. Lifetime for cancer effects.

1. These values were based on professional judgment and are consistent with U.S. EPA Guidance (RAGS).

Dermal Absorption of Chemicals From Sediment. Dermal absorption of contaminants from direct contact with sediment is another potential route of contaminant entry into the body for chemicals which are readily absorbed. This exposure route was quantitatively addressed for children who may play in Killbuck Creek. The following relationship was used to quantify the absorbed dose (AD) of each contaminant:

$$AD = \frac{CS \times CF \times SA \times AF \times ABS \times EF \times ED}{BM \times AT}$$

Variable	Value Used	Rationale
AD = Absorbed Dose (mg/kg-day)		
CS = Chemical Concentration in Sediment (mg/kg)	Lowest value of upper-bound 95% confidence limit (arithmetic mean) or maximum value	- Suggested value.
CF = Conversion Factor (kg/mg)	10^{-6}	
SA[1] = Skin Surface Area Available for Contact (cm^2/event)	5,000 (child)	- 50th Percentile of sum of SA for hands arms and legs for 9 to 10 year old child.
AF = Soil to Skin Adherence Factor (mg/cm^2)	1.45	- Suggested value.
ABS = Absorption Factor (unitless)	Organics[2] Metals[3] 0.001	- Predicted values.
EF[1] = Exposure Frequency (events/year)	35 (child)	- One event per week during 8 non-Winter months.
ED[1] = Exposure Duration (years)	10 (child)	- Occurs from ages 5 to 15.
BM = Body Mass (kg)	30 (child)	- 50th Percentile body mass, for a male child (9 to 10 years old).
AT = Averaging Time (days)	Noncarcinogens - 3650 (child) Carcinogens - 25,550	- Exposure duration for noncarcinogenic effects. Lifetime for cancer effects.

1 These values were based on professional judgement and are consistent with U.S. EPA Guidance (i.e., RAGS).
2 See Table 6-18. Predicted Sediment Chemical Dermal Bioavailabilities.
3 Value based on bioavailability of soil-lead developed from data in Moore, et al., (1980), and Day, et al., (1979).

Dermal Absorption of Contaminants from Surface Water and Groundwater. Contact with surface water may be a potential route of chemical exposure for children who play in Killbuck Creek, while contact with contaminated groundwater may be a potential route of chemical exposure to residents while bathing. The following relationship was used to estimate the absorbed dose from direct contact with surface water or groundwater for children or residents, respectively.

$$AD = \frac{CW \times SA \times PC \times ET \times EF \times ED \times CF}{BM \times AT}$$

	Variable	Value Used	Rationale
AD =	Average Absorbed Dose (mg/kg-day)		
CW =	Contaminant Concentration in Water (mg/L)	Lowest value of upper-bound 95% confidence limit (arithmetic mean) or maximum value	- Suggested value.
SA[1] =	Skin Surface Area (SA) Available for Contact (cm^2/event)	5,000 (child) 19,400 (resident)	- 50th Percentile of sum of SA for hands arms and legs for 9 to 10 year old child. Total SA for average adult resident.
PC=	Dermal Permeability Constant (cm/hr)	1×10^{-4}	- Suggested value.
ET[1] =	Exposure Time (hours/day)	0.5 (child) 0.2 (resident)	- Assume a child wades in Creek for half an hour. Suggested value for shower/bath.
EF[1] =	Exposure Frequency (days/year)	35 (child) 365 (resident)	- One event per week during 8 non-winter months. Bath once daily.
ED[1] =	Exposure Duration (years)	10 (child) 30 (resident)	- Assume children play in creek from age 5 to 15 years. 90% upper-bound estimate of the time a person lives at a single residence.
CF =	Volumetric Conversion Factor For Water (L/cm^3)	0.001	- Suggested value.
BM =	Body Weight (kg)	30 (child) 70 (resident)	- Child's average body mass (i.e., 9-10 years old). Residents average body mass.
AT =	Averaging Time (days)	3650 days (child) or 10,950 (resident) for noncarcinogens and 25,550 for carcinogens (child & resident)	- Exposure duration for noncarcinogenic effects. Lifetime for cancer effects.

1 These values were based on professional judgement and are consistent with U.S. EPA Guidance (RAGS).

Ingestion of Chemicals From Groundwater. The following relationship was used to estimate the CDI of groundwater contaminants by residents via direct ingestion.

$$CDI = \frac{CW \times IR \times EF \times ED}{BM \times AT}$$

Variable	Value Used	Rationale
CDI = Average Chronic daily intake (mg/kg-day)		
CW = Chemical Concentration in Water (mg/L)	Lowest value of upper-bound 95% confidence limit (arithmetic mean) or maximum value	- Suggested value.
IR = Ingestion Rate (liters/day)	2 (resident)	- Suggested value.
EF[1] = Exposure Frequency (days/year)	365 (resident)	- Assumed daily exposure year round.
ED[1] = Exposure Duration (years)	30 (resident)	- 90% upper-bound estimate of the time a person lives at a single residence.
BM = Body Mass (kg)	70 (resident)	- 50th Percentile body mass for adult.
AT = Averaging Time (days)	Noncarcinogens-10,950, Carcinogens -25,550	- Exposure duration for noncarcinogenic effects. Lifetime for cancer effects.

1. These values based on professional judgement and are consistent with U.S. EPA Guidance (RAGS).

6.4 Toxicity Assessment

Toxicological evaluations for most chemicals of potential concern characterizing their inherent toxicity have been performed by the U.S. EPA and have been used to derive toxicity values for use in BRAs. The evaluations consist of a review of scientific data to determine the nature and extent of the human health and environmental hazards associated with exposure to the various chemicals. These characteristics are briefly summarized in Table 6-10 for those individual chemicals which contributed substantially (i.e., greater than 5% for a particular medium) to the total Site risk. These toxicity profiles provide the qualitative weight-of-evidence used to evaluate the actual or potential hazards associated with the chemicals of potential concern. The scientific literature can be reviewed for more complete information regarding the toxicity of the contaminants.

6.4.1 Dose-Response Relationship

Dose-response relationships correlate the magnitude of the chemical intake with the probability of toxic effects. The toxicity produced by a chemical may be categorized as a cancer or noncancer effect. Because the basic biological assumptions applied by the U.S. EPA differ, relating to how chemicals produce cancer vs. how they produce other toxicities (i.e., noncarcinogen effects), the U.S. EPA has developed separate methods for carcinogenic risk estimation and noncarcinogenic risk estimation.

The following describes the distinction between the dose-response for non-cancer and cancer effects:

- Noncarcinogenic Health Effects - Noncarcinogenic health effects may occur upon exposure to a certain dose of a chemical. Noncarcinogenic health effects are viewed as possessing thresholds (i.e., doses which must be exceeded before biological harm is produced). Therefore, a health risk is believed to exist only if established threshold doses are exceeded. The U.S. EPA provides reference dose (RFD) values for evaluating a chemical's noncarcinogenic toxicity.

- Carcinogenic Health Effects - Exposure to a carcinogen could potentially be associated with adverse health implications (cancer). In contrast to noncarcinogenic effects, cancer in theory may occur due to any level of chemical exposure (i.e., no threshold dose). Therefore, the risk (i.e., probability) of contracting cancer increases with increasing levels of exposure. For those compounds judged to be potentially carcinogenic in humans, models are used to predict the relationship between cancer risk and low exposure dose levels. Two models are primarily used by the U.S. EPA: the linear extrapolation model and the linearized multi-stage extrapolation model. Both models produce a straight line relationship at low dose levels, the slope of which is termed the slope factor (SF).

- Reference doses and cancer slope factor values are established by the U.S. EPA. The RAGS encourages use of these values as a consistent tool to compare the hazard associated with exposures to different chemicals at a Site, as well as to have a consistent base by which to compare the hazards from Site to Site. Available (published) reference doses and slope factors for the chemicals of potential concern are listed in Table 6-11.

6.4.1.1 Noncarcinogens. The toxicity values for noncarcinogens are described by a RFD exposure value. The RFD values represent an estimate of an exposure level not anticipated to cause an adverse effect over a set period of exposure. Reference doses are estimated for both subchronic and chronic lengths of exposure. A chronic length of exposure is defined by convention, to be ten percent of a lifetime or longer (i.e., more than 7 years). Shorter periods of exposure are defined as subchronic lengths of exposure. Use of this information allows for characterization of noncarcinogenic health risks.

RFD values are derived from toxicity studies (usually animal studies or human epidemiological studies, if available) on the relationship between chemical exposure and noncarcinogenic toxic effects. They are designed to be protective of sensitive populations and are route specific (e.g., oral or inhalation).

The RFD is expressed as the milligrams (mg) of the chemical administered per unit body weight (kg) per unit time (day). The RFD is generally derived from a no-observed-adverse-effect-level (NOAEL) or a lowest-observed-adverse-effect-level (LOAEL). Uncertainty factors are applied to NOAELs and LOAELs to arrive at the RFD value. An uncertainty factor of 10 is used when applying the results of animal studies to humans. An additional factor of 10 is applied to account for intraspecies variability (i.e., differences between individuals of the human population). An additional factor of 10 is applied if subchronic studies are used in place of chronic studies to estimate a chronic RFD.

The RFD values provided by the U.S. EPA are route specific. Values for inhalation and oral routes of exposure are provided; however, several of these values are not available for the chemicals of potential concern at the WRL Site. Also, there are no values available for the assessment of dermal chemical exposure.

For this assessment, the oral toxicity factors for each compound were used in conjunction with oral bioavailability information to estimate dermal toxicity values for each chemical (Table 6-11). By adjusting the oral reference doses for the fraction of the chemical which is actually absorbed from the oral route, the RFD was converted to the absorbed dose of the chemical which would be considered safe. This is consistent with guidance presented in RAGS Appendix A for adjustment of critical toxicity values. An estimate of the oral bioavailability of each contaminant is presented in Table 6-12. The oral bioavailability of a compound is defined as the proportion of the chemical ingested that actually is absorbed into an organism's body.

In the case of sediments, a large proportion of the chemical is bound to the sediments and is not available for absorption into the body. A chemical must be released from sediment before it can be potentially available for absorption into the body. Therefore, the chemical adsorption potential for each contaminant was used to estimate the amount of each chemical that may be soluble in sweat (i.e., vehicle for contaminant transport to the skin surface). It was assumed that 100 percent of the chemical that is soluble in sweat is bioavailable for absorption into the body. This assumption may overestimate the CDI due to dermal absorption (i.e., 100% absorption) for some chemicals, and therefore represents a worst case estimate. Based on the limited data on the absorption of organic chemicals by the dermal route, this worst case assumption seemed prudent.

6.4.1.2 Carcinogens. The SF is used to estimate the lifetime (assumed to be 70-years) probability of humans contracting cancer as a result of exposure to known or suspected carcinogens. This index is generally reported in units of the inverse of the amount of chemical intake (mg) per unit body weight (kg) per unit time (day) and is derived through an assumed low-dosage linear relationship to effect an extrapolation from high to low dose-responses determined from animal studies. The value used in reporting the slope factor is the upper-bound 95 percent confidence limit of the slope of the dose-response curve. The upper-bound 95 percent confidence limit is a statistical value which means there is a 95 percent chance that the value of concern (i.e., SF) is no greater than what it was estimated to be.

Like reference doses, SFs provided by the U.S. EPA are route specific. Values for inhalation and oral routes of exposure are provided; however, several of these values are not available for the chemicals of concern at the WRL Site. Also, there are no values available for the assessment of dermal chemical exposure; therefore, the oral bioavailability of each compound was used to adjust the SF to account for the percentage of the chemical that is absorbed (i.e., absorbed fraction vs. total intake).

6.5 Risk Characterization

In this section of the report, the potential risks to public health associated with chemical contamination of the WRL Site are presented.

6.5.1 Procedures Used to Calculate Health Risk

Health risks in this assessment were evaluated for noncarcinogenic and carcinogenic effects of the chemicals of potential concern. Cancer was assumed to be the health effect of most concern, but health risks due to noncancer effects were also estimated for potentially carcinogenic compounds if reference doses were available.

Noncarcinogens. Estimating the risk due to a chemical eliciting a noncarcinogenic effect is accomplished by calculating a hazard quotient (HQ). The HQ is a unitless parameter obtained by dividing the estimated Site-specific chronic daily intake (i.e., CDI) or absorbed dose (AD) of the chemical by the available reference dose (RFD) for that chemical. The RFD used must be based on the administered intake of the chemical by the test species (e.g., rat) if a chronic daily intake is used to estimate the magnitude of chemical exposure. The RFD used must also be based on the absorbed fraction of the chemical which was administered to the test species (e.g., rat) if an AD is used to estimate the magnitude of chemical exposure (see Table 6-11). The following equation was used to calculate HQs:

$$\text{Hazard Quotient (HQ)} = \frac{\text{Estimated CDI or AD (mg/kg/d)}}{\text{Reference Dose (mg/kg/d)}}$$

If the HQ exceeds unity, there is a potential health risk associated with exposure to the particular chemical evaluated (U.S. EPA, December 1989). The HQ is not a mathematical prediction of the severity of toxic effects; it is simply a numerical indicator of the transition from acceptable to unacceptable levels of chemical intake.

To predict levels of health risk associated with multiple chemical exposures, the HQ for each chemical is summed to estimate the cumulative noncancer risk. This value is defined as a hazard index (HI). If the HI exceeds unity, there is the potential for noncancer type effects occurring in the population exposed to the chemical mixture.

Carcinogens. Carcinogenic risks (risk) can be estimated by multiplying the cancer slope factor (SF) for a compound by the estimate of an individual's chronic daily intake or absorbed dose of the chemical. Similar to the calculation of the HQs, either the chronic daily intake or absorbed dose (see Table 6-11) may be used to estimate the average level of chemical exposure, but the proper SF must be used (i.e., based on administered or absorbed dose, respectively).

Carcinogenic risks were calculated in the following manner:

$$\text{Risk} = (\text{SF})\ (\text{CDI or AD})$$

Where:
- SF = Cancer slope factor $(mg/kg/d)^{-1}$
- CDI = Amount of a contaminant inhaled or ingested by a receptor (mg/kg/day)
- AD = Amount of a contaminant absorbed by a receptor (mg/kg/day)

The resulting risk is a unitless expression of an individual's likelihood of developing cancer within his/her lifetime as a result of exposure to the carcinogenic indicator chemicals. This likelihood is in addition to the risks incurred by everyday activities. The risk (e.g., 1×10^{-6} or a 1 in 1,000,000 chance) can also be applied to a given population to determine the number of excess cases of cancer that could be expected to result from exposure (e.g., 1×10^{-6} is one additional case of cancer in 1,000,000 exposed persons).

Total risks for exposure to multiple compounds can be presented as the summation of the risks for individual chemical intakes, assuming that there are not antagonistic/synergistic effects between chemicals and that all chemicals produce the same result (cancer). Cancer risks from various exposure routes are also additive, if the exposed populations are the same.

Carcinogenic risks are calculated using a number of assumptions, and many uncertainties are introduced into the values. Factors limiting the extent to which the human health risks can be characterized are primarily associated with the estimation of toxicity and include various uncertainties in the toxicological data base. Extrapolation of non-threshold (carcinogenic) effects from high to low dose, variance in endpoints used for determination of potential health effects, extrapolation of the results of animal studies to human receptors, and varying sensitivity between individuals are examples of uncertainties which make definitive characterization of health risks infeasible.

6.5.2. Superfund Health Risk Goals

Under U.S. EPA guidance (NCP-40 CFR 300.430(e) 1990), when applicable or relevant and appropriate requirements are not available for carcinogenic chemicals of potential concern at a Site, remedies considered should reduce ambient chemical concentrations to levels associated with a carcinogenic risk range of 1×10^{-4} to 1×10^{-6}, where possible.

For noncarcinogens, the U.S. EPA recommends that exposure point concentrations should be reduced to correspond to acceptable chronic daily intake levels (i.e., HQ ≤ 1). If the HQ is less than 1, persons potentially exposed to Site contamination would not be expected to be harmed, based on the assumed exposure conditions. As mentioned above, it is assumed that a person must ingest an amount of chemical above a certain threshold dose before the chemical will cause a noncarcinogenic toxic effect. If the HQ is less than 1, the dose of chemical is below the chemical's threshold dose. For multiple chemical exposures, the HI should be less than 1, so the combined dose of chemicals does not reach the chemical mixture threshold dose.

6.5.3 Public Health Risk Evaluation

The goal of the Public Health Risk Evaluation is to evaluate the likelihood that harm will result from specific chemical exposures. Exposure estimates (i.e., CDI or AD) are combined with toxicity information (i.e., RFD or SF) to estimate the potential for noncancer effects and the risk of developing cancer from human chemical exposures.

As described in the previous section, hazard indices are used to estimate the likelihood that noncancer effects may occur from exposure to mixtures of chemicals, while cancer risk estimates are used as an estimate of the risk of contracting cancer from the same chemical exposures.

Individual chemical hazard quotients (HQs) and cancer risks based on current and probable future land use conditions are summarized by exposure pathway in Tables 6-19 through 6-22, respectively. Risks were calculated based on estimated 95% UBCL or maximum chemical concentrations detected at the WRL Site to represent reasonable worst case exposure conditions. These values are organized according to exposure pathways (e.g., groundwater exposure). A cumulative risk for each pathway is calculated by summing the HQ for noncarcinogens (called the hazard index HI) or the cancer risks for carcinogens for each of the chemicals for which a person is exposed within a specific exposure pathway.

In Section 6.2, chemicals of potential concern were identified based on specific criteria outlined in RAGS. As noted in Section 6.2, contamination near the WRL Site, particularly in groundwater, may be attributed to the WRL Site and other off-site contaminant sources. The relative contribution of the Site and off-site sources to total contamination for each medium could not be quantitated. However, the RI recognized differences in potential source characteristics of contamination between potential upgradient VOC sources and the largely inorganic (i.e., chloride) based plume from the WRL Site. Similarly, the relative contributions of the WRL Site and other potential off-site contaminant sources to the total Site risk are not quantitated in this assessment. This assessment does not separate the effects or risks of individual contamination sources, it simply assesses risks based on the values of contaminants detected in various media around the WRL Site. This public health risk evaluation calculates health risk estimates for each chemical for which U.S. EPA verified toxicity values (i.e., RFD or SF) were available to arrive at a total Site risk for specific exposure pathways (e.g., groundwater exposure).

6.5.3.1 Current Site Conditions. Exposure of children to chemicals in Killbuck Creek sediment and surface water was considered to be the most realistic scenario for exposure to contaminants identified at the WRL Site under current conditions. Based on analytical results, private drinking water downgradient of the Site does not appear to be affected by the landfill; therefore, this medium was not considered a source of chemical exposure under current conditions. Under the assumptions made (i.e., children playing in the Creek 1 day/week, 8 months/year for 10 years), potential noncarcinogenic hazards and potential cancer risks related to sediment and surface water exposure were estimated.

The cumulative HI due to exposure to sediment via both incidental ingestion and dermal absorption and surface water by dermal absorption was $1x10^{-2}$, based on reasonable maximum exposures to noncarcinogenic chemicals present in Killbuck Creek.

It is evident from these risk calculations that exposure to the noncarcinogenic contaminants in surface water and sediment are not expected to cause systemic health effects. In addition, these risk calculations likely overestimate risk, because they were based on the maximum concentration of noncarcinogenic contaminants in sediment, with the exception of barium, which was based on an estimate of the 95% UBCL.

The cumulative cancer risks for the same pathway were calculated to be $6x10^{-7}$ based on the reasonable maximum exposures to carcinogenic chemicals detected in Killbuck Creek. Again, the risks calculated are conservatively high, because they were based on the maximum concentration of carcinogenic contaminants detected in surface water and sediment. This level of cancer risk falls below the U.S. EPA's target risk range for remediation at CERCLA Sites.

6.5.3.2 Future Site Conditions. To assess the risks due to the potential future use of the area surrounding the Site, the risks associated with groundwater exposure were assessed. Noncancer and cancer risks are summarized in Tables 6-21 and 6-22 for groundwater exposure.

To evaluate the potential risk via groundwater consumption, the present concentrations of contaminants in the wells (i.e., water table and piezometers) on-Site and downgradient of the WRL Site were used to estimate the exposure point concentrations of the chemicals. It was assumed that persons were exposed to the contaminated water daily for 30 years and that exposure occurred as a result of groundwater ingestion, as well as dermal contact and inhalation while bathing.

For this exposure scenario to be applicable, persons must inhabit the property directly adjacent to the landfill and use the shallow groundwater as a source of drinking water. Alternately, the plume of contamination would have to migrate approximately 2,000 ft downgradient from its present location, and affect private wells for a prolonged period of time (i.e., 30 years).

Groundwater Noncancer Health Risk Estimates for Chemicals Detected at the WRL Site. The cumulative HI due to exposure to chemicals of potential concern in groundwater via ingestion, dermal absorption, and inhalation was HI = 5 based on reasonable maximum exposures to noncarcinogenic contaminants.[1] Based on these risk calculations, exposure to noncarcinogens in groundwater at the WRL Site may cause adverse health effects, assuming the hypothetical future exposure conditions occur. The majority (84%) of the noncancer health risk was associated with the potential exposure 1,2-dichloroethene (26%), arsenic (5%), barium (7%), manganese (7%), thallium (22%) and zinc (17%).

1. The cobalt daily intake (0.3 mg/day) provided in Doull et al. 1980 was divided by the average mass of an adult human (70 kg) to convert the estimate to the proper units (mg/kg-day) for comparison with the site specific estimate of the daily intake of cobalt due to potential groundwater exposure. There is a large uncertainty associated with the cobalt reference dose and exposure estimate. The cobalt reference dose is an interim value (see Appendix I) that appears conservatively low in comparison to the normal daily intake of this essential element. The daily intake estimated due to groundwater consumption (8.9e-04 mg/kg-day) is lower than the normal daily intake of cobalt (4.3e-03 mg/kg-day). This indicates the level of potential cobalt exposure may not be harmful to humans. Also, cobalt was detected infrequently (i.e., only in two wells) near the landfill. Due to the fact that the interim oral cobalt RFD appears to be unrealistically low, and cobalt was detected infrequently, inclusion of cobalt in the calculation of the future groundwater risk estimate is not believed to be appropriate.

The noncancer health risk is associated primarily with 1,2-dichloroethene, thallium, and zinc ingestion from groundwater. The HQ for each of these chemicals was equal to 1. These risk estimates indicate noncancer effects may not occur as a result of exposure to these chemicals. As exposure to these chemicals may occur simultaneously, there is a greater potential that the combined exposure to these chemicals may result in noncancer health effects. This would generally occur if the chemicals affect the same target organ. In this case, 1,2-dichloroethene and zinc act on the same organ system (i.e., blood) to elicit their respective toxic effects. Thallium works on other organ systems than blood (e.g., hair follicles). For this reason, 1,2-dichloroethene and zinc exposure in combination may pose a health risk at the WRL, but a certain conclusion cannot be drawn because of the limited toxicological information on the interactions of these chemicals when ingested in combination. Nonetheless, the potential for noncancer effects would be low based on the small calculated HI.

Groundwater Cancer Health Risk Estimates for Chemicals Detected at the WRL Site. The cumulative cancer risk for the same pathway was calculated to be $1x10^{-3}$ based on the reasonable maximum exposures to carcinogenic contaminants in groundwater at the WRL Site. The majority (91%) of the cancer health risk was associated with the potential exposure to vinyl chloride (74%) and arsenic (17%). Each of these chemicals is associated with both the WRL Site and off-site upgradient sources of contamination.

Applicability of the Future Groundwater Use Scenario. Although the preceding risk estimates address groundwater exposure under potential future land use conditions, they do not address the likelihood that residents will actually be exposed to contaminated groundwater. The majority of the cancer risk (91%) associated with groundwater contamination is related to a very mobile contaminant and a less mobile contaminant (i.e., vinyl chloride (74%) and arsenic (17%), respectively). The majority of the groundwater risk due to noncancer effects is associated with less mobile contaminants (i.e., metals). Tables 6-21 and 6-22 indicate the relative contribution of each chemical to the total noncancer health effects and cancer risks at the Site.

In general, arsenic and other metals will migrate only a short distance from their source, because of adsorption and precipitation of the metals in the aquifer. Unlike the water solubility of organic chemicals, the water solubility of a metal may be affected by reducing conditions and high salt concentrations caused by the influx of leachate from the landfill into the groundwater. The influx of leachate into groundwater was traced using both chloride and the relative amounts of sodium + potassium, magnesium, and calcium concentrations in the groundwater. Based on the groundwater results, the elevated levels of metals were associated with the leachate-affected wells. Levels of metals in other wells appeared to be near natural background metal concentrations in groundwater. This indicates that the migration of the metals is limited to the areas adjacent to the landfill. For example, arsenic in aqueous solutions under oxidizing conditions can precipitate from solution and/or become strongly adsorbed by most soils and sediments, similar to phosphate. However, under reducing conditions, arsenic shows much less affinity for adsorption and is more water soluble, and therefore, a greater portion of the total arsenic is in solution. Localized reducing conditions likely exist in areas of leachate-affected groundwater. Under this scenario, both naturally occurring and leachate arsenic may be solubilized. As oxidizing conditions increase downgradient, aqueous arsenic is precipitated and/or adsorbed out of solution, thus minimizing migration.

Based on the consideration that metals appear to migrate only a short distance from the landfill, the likelihood that residents would be exposed to elevated metal concentrations in their drinking water due to WRL Site contamination in the future would be expected to be low, unless persons developed the area directly adjacent to and downgradient of the landfill. Rather, it is more likely that volatile organic chemicals potentially could migrate further downgradient in the future based on their solubility and low affinity for aquifer materials. The magnitude of this future migration has not presently been estimated. As previously mentioned, it is unlikely that future development will occur adjacent to the landfill in the downgradient direction of groundwater flow. This area contains Killbuck Creek and its associated floodplain. Based on practical considerations and regulatory constraints, it is unlikely that a well would be installed for drinking water purposes in this area.

6.5.3.3 Qualitative Risk Estimates. The risks associated with the following chemicals detected at the WRL Site could not be assessed quantitatively due to the lack of published critical toxicity values (i.e., RFD or SF) from the U.S. EPA. Such values were requested from ECAO (Environmental Criteria and Assessment Office), but toxicity values for the chemicals were not available except for cobalt (see Appendix I). A qualitative assessment has been made of the toxicity of each of these chemicals to put their health risk potential in perspective with other health risks at the WRL Site.

Volatiles	Qualitative Toxicity Assessment
1,3-Dichlorobenzene	Low systemic toxicity to humans based on the toxicity of other isomers. Cancer potency unknown, but considered low based on similar isomers.

Metals	Qualitative Toxicity Assessment
Copper	Essential nutrient with low order of toxicity at low to moderate dose levels.
Iron	Essential nutrient with low order of toxicity at low to moderate dose levels.
Lead	Lead is a toxic heavy metal even at relatively low doses. The drinking water standard for lead is 50 ug/L. The concentration of lead in groundwater was less than 50 ug/L in each well. The average concentration of lead in groundwater was 4.75 ug/L. Therefore, this metal's level in groundwater appears to be acceptable based on applicable health criteria.
Magnesium	Essential nutrient with low order of toxicity at moderate to high doses.
Calcium	Essential nutrient with low order of toxicity at moderate to high doses.
Potassium	Essential nutrient with low order of toxicity at moderate to high doses.
Sodium	Essential nutrient with low order of toxicity at moderate to high doses.

6.5.3.4 On-Site versus Upgradient Off-Site Groundwater Health Risk Estimates

The objective of this section is to compare the health risks associated with the potential for human exposure to groundwater on-Site, and off-site upgradient of groundwater flow. This comparison is important to make because directly upgradient of the WRL Site is the Acme Solvents site which, as previously mentioned, is also a Superfund site with known groundwater contamination.

At the WRL Site, a public health concern is associated with the potential exposure of people to contaminated groundwater in the future. The total noncancer risk estimate was calculated to be 5 and the total cancer risk was calculated to be 1×10^{-3} for the WRL Site.[2] Based on an endangerment assessment conducted by Levine - Fricke for the Acme Solvents Site, the total noncancer risk estimate due to groundwater exposure in the future was 30, and the total cancer risk was 3×10^{-2} (see table 7-2 of Levine - Fricke Report) at the Acme Solvent Site (Levine - Fricke, 1990).

6.5.4 Summary of Potential Health Risks.
The objective of the BRA, as outlined in RAGS, is to characterize the extent of contamination and the potentially exposed population(s) sufficiently to determine what risks need to be prevented.

An exposure assessment was conducted which estimated the exposure potential of area residents, qualitatively or quantitatively, to the following media based on current and potential future land use conditions:

- Air
- Drinking water
- Sediment
- Surface water
- Food

Under the current use scenario, surface water and sediment in Killbuck Creek appear to pose the most likely point of chemical exposure to individuals (children) living in the area of the WRL Site. Based on area census information, several families live near the WRL Site. Although behavioral patterns of individuals in the area of the WRL Site cannot be predicted conclusively, children were considered to represent the most sensitive subpopulation that may become exposed to these media under current conditions.

2. Cobalt was detected in two wells, P1 (84 ug/L) and P4R (63 ug/L in duplicate sample only, not detected in investigative sample) during Round 1 groundwater sampling. Due to the fact that cobalt was detected in a limited number of wells and the interim oral RFD (see Appendix I) appears to be unrealistically low (i.e., below a normal daily human intake of cobalt), inclusion of cobalt in the calculation of the future groundwater risk estimate is not believed to be appropriate. However, if cobalt is included, it would represent 95% of the nancancer risk, resulting in a Hazard Index = 100.

chemicals to the air at the WRL Site. Levels of chemicals detected in air downwind of the WRL Site were not elevated above upwind chemical concentrations. Also, contaminated groundwater downgradient of the WRL Site is not used for drinking water purposes (i.e., no private wells in this area). Finally, food crops for human consumption are not grown at the WRL Site.

In the future, there is the potential for exposure conditions to change if land use practices change. Based upon possible future land use conditions, persons may use the groundwater near the WRL Site as a source of drinking water. Although this route of exposure is possible, it assumes people will live adjacent to the WRL Site, or the contaminated groundwater plume will move downgradient of the WRL Site to private well locations. Both of these assumptions are unlikely, based on regulatory restraints associated with placing drinking water wells in areas of known groundwater contamination and the limited amount of contaminant movement in groundwater that has been observed during the RI.

Also under future use considerations, exposures of humans to other media (i.e., air, food) were considered, but were not expected to contribute substantially to human chemical exposure. In the future, the probable land use scenario is closure of the WRL Site. Air releases are expected to be minimal, because the gas collection system is expected to be maintained to collect and destroy landfill gas. In addition, cover material is anticipated to limit chemical release to the atmosphere. Because of the nature of Site contamination (primarily low concentrations of VOCs in groundwater), crops grown in the area are not likely to be affected in the future.

Combining the potential for human exposure to chemicals in Site media with toxicity (both noncancer and cancer) information for chemicals of potential concern, risk estimates were calculated based upon current and possible future Site conditions.

Based on current Site conditions, it was assumed children will play in Killbuck Creek and may be exposed to sediment by incidental ingestion and contact, and to surface water via direct contact. Assuming these exposure conditions, noncarcinogenic health effects are not expected (i.e., $HI < 1$) and cancer risks are low (i.e., $< 1x10^{-6}$).

Under a hypothetical future use scenario, it was assumed that residents would be exposed via ingestion, as well as dermal contact and inhalation, to the contaminants in groundwater. Unlike under current Site conditions, noncarcinogenic health effects may be of concern and cancer risks are substantially greater than the U.S. EPA's risk range ($1x10^{-4}$ to $1x10^{-6}$), assuming residents were exposed daily to the chemicals of potential concern in groundwater for 30 years.

The primary noncarcinogenic health risk (84%) was associated with potential groundwater exposure to 1,2-dichloroethene (26%), arsenic (5%), barium (7%), manganese (7%), thallium (22%), and zinc (17%). The noncancer risk (i.e., HQ) estimate associated with 1,2-dichloroethene, thallium, and zinc consumption from groundwater were each equal to one. Interactive effects are possible for 1,2-dichloroethene and zinc, therefore, the combined exposure to these chemicals may pose a low potential for noncarcinogenic blood effects (e.g, anemia) to occur in humans.

On the other hand, the majority (91%) of the cancer risks were due to exposure to arsenic (17%) and vinyl chloride (74%). Both chemicals are known human carcinogenics.

Currently, the WRL Site does not appear to present a public health concern. However, potential future exposure to groundwater at the WRL Site appears to present a potential public health concern based upon estimates of noncancer and cancer risk. Although groundwater exposure at the WRL Site is above U.S. EPA Risk Goals, the health risk at the WRL Site is similar to anthropogenic levels of risk associated with the off-Site upgradient groundwater contamination. Therefore, although future potential exposure to groundwater at and downgradient of the WRL Site appears to be a potential public health concern, anthropogenic background appears to contribute to this risk.

6.5.5 Uncertainty in the Health Risk Evaluation

There are several areas of inherent uncertainty in this evaluation of public health risk at the WRL Site including:

- Quantification of CDIs for each receptor population
- Extrapolation of high dose animal toxicological data to low dose human exposure health risks

- The use of chemicals of potential concern to represent total Site risk (i.e., inability to include all chemicals of potential concern identified in the assessment, because of lack of toxicology data)
- Only limited potential contaminants were analyzed at the Site (e.g., the numerous TICs detected on-Site)

With respect to uncertainties associated with quantification of CDIs, assumed behavior patterns contain inherent uncertainties. Because it would be impossible to model specific behavior patterns of the individuals around this Site, certain representative population behavioral assumptions had to be made. In general, these behavioral assumptions are conservative, leading to over-estimations of health risks for each receptor population. Some of the more conservative behavior assumptions in this assessment include:

- Individuals residing near the WRL Site will live in the area for 30 years.
- Children who play in Killbuck Creek come into contact with surface water and sediment at the frequency and magnitude described in the Exposure Assessment section.
- The hypothesized future development near the Site occurs and exposes residents at the frequency and magnitude described in the Exposure Assessment section.

Several uncertainty factors may be associated with the use of data as a result of sample analyses and validation of the data. Data with estimated concentrations of compounds (data flagged with "J") were used in the assessment, even though a certain amount of error may be associated with the estimated concentrations, the result of which may be concentrations greater or less than the reported concentrations.

Animal to human extrapolation of toxicological data, as well as high dose to low dose extrapolation, also contain considerable uncertainty. This uncertainty is inherent in the various RFD and SF values used in this assessment and usually leads to an overestimation (conservative) of the toxicity of chemicals at low dose levels.

An additional assumption is used to evaluate exposure to multiple carcinogens or multiple noncarcinogens. Both the possibility of synergism between chemicals and the possibility of no toxicological interaction between chemicals exist. The assumption of an additive effect between chemicals is an attempt to be conservative in this assessment, but leads to additional uncertainty with respect to specific chemical mixtures.

Because the final health risk estimates at the WRL Site are generated by comparing the estimated exposure doses (with their uncertainties) with appropriate reference doses (with their uncertainties), the final risk estimates should be viewed only as approximate. The health risks which have been calculated for the WRL Site should be used more on a relative (or comparison) basis by risk managers, rather than on an actual risk basis.

6.6 Environmental Assessment

The objectives of this component of the Baseline Risk Assessment are to characterize the natural habitats which may be influenced by the Site and to appraise the actual or potential adverse effects contaminants have had on these habitats. Relative to the human health assessment, the methodology for an ecological assessment is much less defined. The Risk Assessment Guidance For Superfund-Environmental Evaluation Manual (EPA, March 1989) has been published by the U.S. EPA as a guide to conducting ecological assessments at Superfund Sites. This guidance document was consulted in assessing the potential ecological impacts of the WRL Site on the surrounding ecology of the area.

The overall approach to the environmental assessment is analogous to that of human health assessments, and includes identifying contaminants of potential concern, pathways of contamination migration, and populations (flora and fauna) potentially affected by Site contamination. To the extent possible, actual adverse impacts to natural habitats are estimated. Similarly, the potential for future environmental impact is also described.

6.6.1 Site Description

Prior to disposal activities, land comprising the waste disposal area at the WRL Site was excavated for its sand and gravel. Thus, natural habitats existing prior to mining operations at the Site were destroyed. At present, the waste disposal area is active and does not support habitat for plants and animals. Land in the vicinity of the WRL Site has been developed for agricultural, residential and commercial purposes. Sensitive ecological habitats (i.e., Killbuck Creek and adjacent wetlands) are located downgradient of the WRL Site. Therefore, migration of Site contaminants via groundwater or surface erosion have the potential to affect these areas.

Killbuck Creek was rated as a B-Stream - Highly Valued Aquatic Resource (Illinois EPA 1989), meaning the Creek is rated as a good fishery for important gamefish species; but species richness may be somewhat below expectations for stream size or geographic region. The Creek has a maximum width of approximately 50 ft and an average depth of 1 to 2 feet. As indicated by the Stream survey, the Creek has the potential for supporting a wide range of fish, benthic invertebrates, and plant life.

Based on a current wetlands inventory (ENCAP 1990), a 3.73-acre area of wetlands exists on-Site. The wetlands are located approximately 150 feet south of the landfill, and extend to the west into the Creek's floodplain. The wetlands are composed of 3.18 acres of scrub-shrub/forested wetland, 0.46 acre of shallow drainage way, and 0.09 acre of farmed wetland. The wetlands support a diverse array of trees, shrubs, and herbaceous species of plants. This area is frequented by wildlife; the most noticeable species include birds, small mammals, and possibly deer. Although an inventory of terrestrial plant and animal species has not been performed, the Site is not known to be inhabited by endangered or threatened species.

6.6.2 Assessment of Ecological Risks- General Theory and Approach

To assess the ecological impact of chemical releases from a Site is a very complex task. If an individual species is adversely affected within an ecosystem, there is a potential for adverse foodchain, foodweb, or community effects within the ecosystem, because other species rely on that affected species for their own needs (e.g., food, shelter). To simplify the task of estimating whether a chemical release may adversely affect an ecosystem, it was assumed that the health of an ecosystem depends upon the health of the individual species within the ecosystem. Thus, if the health of each individual species within the ecosystem can be maintained, there should not be higher level ecosystem effects (e.g., structural changes in communities). From a practical point of view, it is not possible to estimate the effects that a chemical release would have on each individual species within an ecosystem. A more practical approach is to estimate the chemical effect on groups of animals which have similar habitats and eating habits.

The potential for a species to be adversely affected by a chemical release depends on a species' sensitivity to the chemical and the environmental exposure concentration. The species' sensitivity is dependent on the way in which the chemical interacts with the physiology of the species and the species' surroundings. The concentration to which an organism is exposed is dependent on the concentration of the chemical at the point of its release, the routes of chemical transport, and the fate of the chemical in the environment (i.e., the way in which the chemical is eliminated and partitions within media). The highest proportions of many chemicals partition into only a few media (e.g., sediment, air, water) within an ecosystem, because of their physical and chemical properties (i.e., water solubility and vapor pressure). Thus, the greatest chemical exposure occurs to those groups of organisms that reside in the most heavily contaminated media or habitats. Within the wetland and Killbuck Creek ecosystems, the organisms with the greatest potential for chemical exposure are the aquatic species that live their entire lives in the Creek. For this reason, fish and benthic invertebrates were considered the most sensitive groups of organisms in the ecosystems downgradient of the WRL Site.

Within an ecosystem, there is not only chemical transfer from contaminated media to organisms, but also transfer from organism to organism. Thus, the food source of animals must be considered as a potential source of chemical exposure. The plants and animals of an ecosystem can be grouped and ranked according to the type of food that they ingest. The lowest ranked group (i.e., trophic level) is plants that store the energy from the sun. The food that the plants produce is used by animals in higher trophic levels. Plants are called the primary producers. Animals which feed on the primary producers are called primary consumers. An example of a primary consumer would be a rabbit or zooplankton. Animals that feed on primary consumers are called secondary consumers. An example of a secondary consumer is a hawk or a lynx. When species rely on others as a food source, a series called a food chain develops. The series of plants and animals previously described (i.e., grass--rabbit--lynx) is an example of a foodchain. As a result of the animals in the higher trophic levels consuming many organisms in the lower trophic levels, there is the potential for a chemical to concentrate at higher trophic levels. This concentration of chemicals in higher trophic level species sometimes leads to the magnification of the chemical concentration in the tissues of those animals; this process is termed biomagnification. If a chemical is easily absorbed into the bodies of animals, not easily metabolized by the organism, and not excreted effectively, the chemical has the potential to biomagnify through the foodchain. If the chemical biomagnifies, there is the possibility that the chemical exposure will become great enough to cause harm to animals in higher trophic levels (e.g., hawk), even though effects are not present in lower trophic levels (e.g., plants). Thus, the eating habits of an animal can be an important factor in determining its environmental exposure to a chemical in the ecosystem.

In summary, the greatest chemical exposure occurs to those groups of organisms that reside in the most heavily contaminated habitats and that consume contaminated prey (e.g., fish). Thus, it is important to determine the chemical sensitivity of this particular group of organisms and consider their estimated environmental exposure concentration to determine if there is the potential for adverse health effects.

6.6.3 Exposure Assessment

At the WRL Site, the potential exposure to fish is low. Concentrations of chemicals in Killbuck Creek sediment were measured during the RI, and the majority of the chemicals (i.e., inorganics) were detected at natural concentrations at downstream sample locations. Few organic chemicals (see Table 6-1) were detected in downstream sediment samples, and their presence was infrequent. Therefore, exposure to this medium was only considered indirectly to fish via its potential impact on surface water.

The main contamination of the WRL Site is associated with the groundwater, but this does not present a potential direct route of exposure to fish. Groundwater at the WRL Site discharges and flows beneath Killbuck Creek, and there is the potential for contaminated groundwater to discharge to the stream. Also, there is the potential for surface water drainage from the WRL Site to enter Killbuck Creek. Surface soil and floodplain sediment were not analyzed as part of the RI; therefore, the likelihood of this pathway is not known. There were few chemicals detected in surface water during the RI (see Table 6-1).

6.6.4 Ecological Risk Evaluation

The levels of the chemicals detected in surface water were compared to Ambient Water Quality Criteria. These criteria are based on the toxicity data for the most sensitive aquatic organism tested. The following is a comparison of the surface water chemical maximum concentrations detected in Killbuck Creek downstream of the WRL Site with the Ambient Water Quality Criteria.

Killbuck Creek Surface Water Chemical	Maximum Surface Water Concentration ug/L	Ambient Water Quality Criteria Lowest Reported Toxic Chemical Concentration for Freshwater: Acute ug/L	Chronic ug/L
Benzene	1.7	5,300	--
Chloroform	0.3	28,900	1,240
Methylene Chloride	19.1	193,000	--
1,1,1-Trichloroethane	0.7	52,800	--
Trichloroethene	1.1	45,000	--
1,2-Dichloroethene	0.7	135,000	--
		EPA Ambient Water Quality Criteria	
	ug/L	ug/L	ug/L
Barium	100.0	--	--
Cyanide (total)	17.0	22	5.2

-- = Criteria have not been developed

It appears that there is little potential for adverse effects to fish in the aquatic ecosystem due to volatile organic chemical exposure. The lowest reported toxic concentration in any freshwater organism is greater than 1000 times greater than what is present in the surface water of Killbuck Creek. Also, these chemicals are not expected to biomagnify, based on their physical and chemical properties (i.e., they are relatively water soluble), and because fish can metabolize and excrete these chemicals.

Exposure of fish to inorganic chemicals is not expected to cause adverse health effects based on acute exposure criteria. The chronic criteria is marginally exceeded for cyanide, based on the maximum concentration of this chemical in surface water. On the other hand, the average concentration of this chemical in surface water was below its criteria and it was detected at a single downstream location. Based on this information, chronic periods of exposure to inorganic chemicals at the concentrations detected in Killbuck Creek surface water should not cause deleterious health effects to fish.

6.6.5 Conclusions

Fish were considered the most susceptible group of aquatic species to chemical exposure in Killbuck Creek. Effects on fish are not expected based on the chemical concentrations in surface water. Therefore, since this sensitive group of organisms appears to be safe from health effects, other aquatic ecosystem effects are not anticipated.

Health risks to the terrestrial environment could not be compared to applicable criteria because floodplain sediment and surface soil samples were not analyzed as part of the RI. Based on visual observations, signs of impacts on the terrestrial ecosystem were not observed (e.g., stressed vegetation). Also, because of the nature of the contamination at the WRL Site (i.e., groundwater), impacts on the terrestrial ecosystem are not expected.

TABLE 6-1

Organic and Inorganic Analytes Detected

Environmental Medium	Analyte	Analyte Concentration Minimum	Maximum	Background(2)	Number of Locations(1) Sampled for Analysis Total	Positive Detection
GROUNDWATER						
Monitoring Wells	Indicators	ug/L	ug/L	ug/L		
	Nitrate + Nitrite	30.0	11,600.0	NA	15	15
	Metals					
	Arsenic	2.0	46.0	13.0	30	9
	Barium	19.0	1,145.0	1,640.0	30	30
	Cadmium	0.2	16.0	2.0	30	21
	Calcium	46,200.0	225,000.0	52,200.0	14	14
	Chromium, Total (3)	0.3	3.5	ND	19	14
	Cobalt	84.0	84.0	ND	14	1
	Copper	122.0	122.0	ND	14	1
	Iron	109.0	11,000.0	219.0	14	8
	Lead	6.0	32.0	ND	14	3
	Magnesium	25,800.0	107,000.0	53,300.0	14	14
	Manganese	41.0	2,010.0	16.0	14	11
	Nickel	46.0	224.0	ND	14	2
	Potassium	9,000.0	141,000.0	ND	14	6
	Silver	2.0	3.0	ND	14	3
	Sodium	6,700.0	280,000.0	18,300.0	14	12
	Thallium	2.0	6.0	ND	14	14
	Vanadium	50.0	60.0	ND	14	2
	Zinc	37.0	6,340.0	763.0	14	12
	Cyanide, Total	6.0	494.0	10.0	19	13
	Pesticides/PCBs	ND	ND	ND	14	0
	Semi-Volatiles					
	1,4-Dichlorobenzene	0.9	63.0	ND	19	15
	1,2-Dichlorobenzene	0.1	7.4	ND		11
	1,3 Dichlorobenzene	0.4	0.4	ND		1
	Acenaphthene	0.6	0.6	ND		1
	Dibenzofuran	0.3	0.3	ND		1
	Diethylphthalate	4.0	4.0	ND		1
	bis(2-Ethylhexyl)phthalate	5.0	36.0	ND		5
	Volatiles					
	Acetone	6.0	11.0	21.0	30	3
	Chloromethane	4.0	4.0	5.0		2
	Vinyl Chloride	0.4	98.0	3.0		16

Environmental Medium	Analyte	Analyte Concentration Minimum ug/L	Maximum ug/L	Background(2) ug/L	Number of Locations(1) Sampled for Analysis Total	Positive Detection
	Chloroethane	0.5	150.0	4.3		18
	Methylene Chloride	1.0	20.0	9.4		7
	1,1-Dichloroethene	0.1	3.5	6.0		8
	Total 1,2-Dichloroethene	1.0	160.0	23.0		16
	trans-1,2-Dichloroethene	0.2	6.5	0.7		13
	cis-1,2-Dichloroethene	0.2	280.0	1200.0		22
	Chloroform	11.0	11.0	2.0		1
	1,1-Dichloroethane	0.1	110.0	190.0		21
	1,2-Dichloroethane	0.2	4.1	13.0		13
	1,1,1-Trichloroethane	0.2	33.0	350.0		15
	1,2-Dichloropropane	0.5	11.0	8.9		15
	Trichloroethene	0.2	160.0	380.0		21
	Dibromochloromethane	0.2	0.4	ND		3
	Benzene	0.5	17.0	3.6		15
	Trans-1,3-Dichloropropene	0.4	2.8	0.7		4
	Bromoform	0.5	0.5	ND		1
	Tetrachloroethene	0.3	75.0	810.0		19
	1,1,2,2-Tetrachloroethane	4.5	18.9	5.7		2
	Toluene	0.2	3.0	5.0		4
	Chlorobenzene	0.3	8.3	ND		13
	Ethylbenzene	0.2	9.0	11.0		14
	Total Xylenes	1.0	50.0	ND		2
	m and p-Xylene	1.0	4.4	9.7		5
	o-Xylene	0.4	6.1	24.0		8
	Carbon Tetrachloride	0.2	8.0	ND		6
	Bromodichloromethane	0.2	0.2	ND		1

Environmental Medium	Analyte	Analyte Concentration Minimum	Maximum	Background(2)	Number of Locations(1) Sampled for Analysis Total	Positive Detection
LEACHATE						
	Indicators	ug/L	ug/L	ug/L		
	Nitrate + Nitrite	200.0	800.0	NA	6	6
	Metals					
	Aluminum	320.0	123,000.0	NA	10	10
	Antimony	11.0	47.2	NA		4
	Arsenic	8.0	318	NA	19	19
	Barium	78.0	4,710	NA		19
	Beryllium	0.3	7.7	NA	10	2
	Cadmium	1.0	266.0	NA	19	17
	Calcium	29,900.0	241,000.0	NA	10	10
	Chromium, Total(3)	143.0	933	NA		10
	Cobalt	56.0	154	NA		8
	Copper	25.0	5,720.0	NA		8
	Iron	420.0	263,000.0	NA		10
	Lead	26.0	1,450	NA		10
	Magnesium	30,800.0	812,000.0	NA		10
	Manganese	37.0	4,110.0	NA		10
	Mercury	0.5	5.9	NA		7
	Nickel	323.3	1,130.0	NA		10
	Potassium	608,000.0	1,750,000.0	NA		10
	Selenium	11.1	12.0	NA		2
	Silver	1.0	21.0	NA		5
	Sodium	10,200.0	3,100,000.0	NA		10
	Thallium	45.5	45.5	NA		1
	Vanadium	13.5	303.0	NA		7
	Zinc	191.00	15,400.0	NA		10
	Cyanide, Total	38.0	6,000.0	NA		10
	Pesticides/PCBs					
	Alpha-BHC	0.059	0.059	NA	10	1
	Beta-BHC	0.065	0.110	NA		2
	Delta-BHC	0.054	0.054	NA		1
	Gamma-BHC (Lindane)	0.086	0.086	NA		1
	Aldrin	0.085	0.720	NA		2
	Heptachlor Epoxide	0.160	0.160	NA		1
	Endrin	0.130	0.130	NA		1
	Endosulfan Sulfate	0.120	0.380	NA		2
	Gamma-Chlordane	0.066	0.092	NA		2
	AROCLOR-1242	2.700	6.900	NA		6
	AROCLOR-1248	7.200	7.200	NA		1
	AROCLOR-1254	1.800	3.800	NA		3
	AROCLOR-1260	1.500	1.800	NA		2
	Semi-Volatiles					
	Phenol	140.0	140.0	NA	10	1
	1,3-Dichlorobenzene	19.0	19.0	NA		1
	1,4-Dichlorobenzene	22.0	27.0	NA		2
	2-Methylphenol	27.0	140.0	NA		3
	4-Methylphenol	30.0	200.0	NA		2
	2,4-Dimethylphenol	33.0	310.0	NA		6

Environmental Medium	Analyte	Analyte Concentration Minimum	Maximum	Background(2)	Number of Locations(1) Sampled for Analysis Total	Positive Detection
		ug/L	ug/L	ug/L		
	Benzoic acid	1,200.0	1,200.0	NA		1
	Naphthalene	6.0	50.0	NA	10	6
	2-Methylnaphthalene	8.0	23.0	NA		2
	Dibenzofuran	11.0	11.0	NA		1
	Fluorene	17.0	17.0	NA		1
	Phenanthrene	6.0	53.0	NA		3
	Anthracene	2.0	2.0	NA		1
	Fluoranthene	12.0	22.0	NA		2
	Pyrene	9.0	9.0	NA		1
	bis(2-Ethylhexyl)phthalate	80.0	1,200.0	NA		6
	Di-n-octylphthalate	13.0	170.0	NA		4
	Volatiles					
	Chloromethane	2.0	15.0	NA	19	1
	Vinyl Chloride	2.0	270.0	NA		8
	Chloroethane	9.0	11.0	NA		1
	Methylene Chloride	10.0	100.0	NA		1
	Acetone	76.0	18,000.0	NA		2
	Carbon Disulfide	1.0	1.0	NA		1
	1,1-Dichloroethene	1.0	38.0	NA		2
	1,1-Dichloroethane	27.0	57.0	NA		2
	Total 1,2-Dichloroethene	4.30	2,100.0	NA		7
	Chloroform	12.0	16.0	NA		2
	2-Butanone	11.0	22,000.0	NA		7
	1,1,1-Trichloroethane	1.10	1.1	NA		1
	1,2-Dichloropropane	0.23	34.0	NA		6
	Trichloroethene	1.0	29.0	NA		2
	Benzene	1.0	7.6	NA		12
	4-Methyl-2-Pentanone	43.0	1,600.0	NA		2
	2-Hexanone	39.0	260.0	NA		2
	Tetrachloroethene	0.7	17.0	NA		2
	Toluene	18.0	730.0	NA		17
	Chlorobenzene	0.3	5.0	NA		7
	Ethylbenzene	1.0	77.0	NA		14
	Styrene	0.61	10.0	NA		10
	Total Xylenes	69.0	300.0	NA		9
	trans-1,2-Dichloroethene	49.0	49.0	NA		1
	cis-1,2-Dichloroethene	0.32	68.0	NA		5
	1,4-Dichlorobenzene	5.4	30.0	NA		7
	m and p-Xylene	1.6	103.0	NA		10
	o-Xylene	2.7	62.0	NA		10
	1,2-Dichlorobenzene	0.32	2.6	NA		4

Environmental Medium	Analyte	Analyte Concentration Minimum	Maximum	Background(2)	Number of Locations(1) Sampled for Analysis Total	Positive Detection
SURFACE WATER						
Killbuck Creek	Indicators	ug/L	ug/L	ug/L		
		NA	NA	NA		
	Metals					
	Arsenic	U10.0	U10.0	3.4	4	0
	Barium	70.0	100.0	81.4		4
	Cyanide, Total	17.0	17.0	U10.0		1
	Pesticides/PCBs	ND	ND	ND	4	0
	Semivolatiles	ND	ND	ND	4	0
	Volatiles					
	Methylene Chloride	19.1	19.1	U5	4	1
	Chloroform	0.29	0.29	U5		1
	1,1,1-Trichloroethane	0.7	0.7	U5		1
	Trichloroethene	1.06	1.06	U5		1
	Benzene	1.74	1.74	U5		1
	cis-1,2-Dichloroethene	0.67	0.67	U5		1

Environmental Medium	Analyte	Analyte Concentration Minimum	Maximum	Background(2)	Number of Locations(1) Sampled for Analysis Total	Positive Detection
SEDIMENT		ug/kg	ug/kg	ug/kg		
Killbuck Creek	Indicators	NA	NA	NA	NA	NA
	Metals					
	Arsenic	700.0	5,900.0	2,200.0	4	4
	Barium	15,200.0	81,900.0	73,400.0		4
	Cadmium	800.0	1,000.0	U5		2
	Calcium	20,400,000.0	54,800,000.0	26,300,000.0		4
	Chromium, Total(3)	2,600.0	8,800.0	5,800.0		4
	Copper	5,500.0	7,600.0	6,900.0		3
	Iron	2,740,000.0	10,800,000.0	9,550,000.0		4
	Lead	1,600.0	7,400.0	8,400.0		4
	Magnesium	10,100,000.0	23,700,000.0	11,600,000.0		4
	Manganese	75,200.0	239,000.0	282,000.0		4
	Nickel	9,100.0	9,800.0	U40		2
	Silver	100.0	100.0	100.0		4
	Thallium	200.0	600.0	200.0		4
	Vanadium	6,600.0	16,700.0	8,800.0		4
	Zinc	8,900.0	27,200.0	34,500.0		4
	Cyanide total	3,400.0	17,500.0	28,600.0		4
	Pesticides/PCBs	ND	ND	ND	4	0
	Semi-Volatiles					
	Di-n-butylphthalate	3,900.0	4,400.0	U10.0	4	2
	bis(2-Ethylhexyl)phthalate	63.0	63.0	U10.0		1
	Fluoranthene	U10.0	U10.0	160.0		0
	Pyrene	U10.0	U10.0	160.0		0
	Benzo(a)anthracene	U10.0	U10.0	84.0		0
	Chrysene	U10.0	U10.0	95.0		0
	Benzo(b)fluroanthene	U10.0	U10.0	95.0		0
	Benzo(k)fluroanthene	U10.0	U10.0	85.0		0
	Benzo(a)pyrene	U10.0	U10.0	95.0		0
	Volatiles					
	Chloroform	2.0	2.0	U5	4	2

NOTES

(1) The total number of positive detects refers to the number of locations sampled where a particular chemical was detected. At a given location, a chemical may have been detected more than once, but this value is not a count of each detect, but rather the number of locations where the chemical was detected. The total number of sample locations represents the total number of leachate samples, on-Site wells, or downstream stations where samples were collected. Sampling stations one through four were considered downstream sample locations based on water flow.

(2) All wells located east of Lindenwood Road were used to represent off-site upgradient groundwater quality. Sampling station five is upstream of the WRL and considered a background sample location.

(3) Chromium risks were estimated assuming the valence state of the element is +3 in each medium. Based on the probable groundwater and sediment conditions near the Site (i.e., reducing or mild oxidizing conditions), chromium would exist in the +3 state.

LEGEND

ND = all chemicals within the specific group were not detected.

U10 = undetected below a specific contract required quantitation limit (CRQL) for organic analytes and below the contract required detection limit (CRDL) for inorganics.

R = data unusable based on data validation review.

NA = Not applicable because no sample was taken or specific parameter was not analyzed.

MWK/vlr/TJD/SGW2/JAH/JFK
[jlv-403-93a]
13160.40-MD

Table 6-2

EXAMPLES OF CONDITIONS WHICH REQUIRE ESTIMATION ("J" DATA VALIDATION QUALIFIER) OF ORGANICS ANALYSIS DATA

1. If the contract-required time period from the time of sampling to the time of sample extraction or analysis (holding time) is exceeded.

2. If the instrument initial or continuing calibration criteria are not within U.S. EPA established limits.

3. If the recoveries of the sample surrogate standards do not meet U.S. EPA established criteria.

4. If the response of the sample internal standards do not meet U.S. EPA established criteria.

5. If the concentration of the compound exceeds the calibration range of the instrument.

6. If the concentration of the compound is below the contract-required quantitation limit.

7. If the compound is a Tentatively Identified Compound.

NOTE

Refer to Appendices for definitions of laboratory and data validation qualifiers.

SOURCE

Laboratory Data Validation Functional Guidelines for Evaluating Organics Analyses, (U.S. EPA, February 1988).

MWK/jkk/BJC/JAH/JFK
[jkk-400-81]
13160.40

TABLE 6-3

Selection of Chemicals of Potential Concern(1)

Chemical	Presence In Media			
	Leachate	Groundwater	Surface Water	Sediment
VOLATILES				
Acetone	X	X/Y	0	0
Benzene	X	X/Y	X/Y	0
Bromodichloromethane	0	X/Y	0	0
Bromoform	0	X/Y	0	0
2-Butanone	X	0	0	0
Carbon Disulfide	X	0	0	0
Carbon Tetrachloride	0	X/Y	0	0
Chlorobenzene	X	X/Y	0	0
Chloroethane	X	X/Y	0	0
Chloromethane	X	X/Y	0	0
Chloroform	X	X/Y	X/Y	X/Y
Dibromochloromethane	0	X/Y	0	0
1,1-Dichloroethane	X	X/Y	0	0
1,2-Dichloroethane	0	X/Y	0	0
1,1-Dichloroethene	X	X/Y	0	0
1,2-cis and trans Dichloroethene	X	X/Y	X/Y	0
1,2-Dichloropropane	X	X/Y	0	0
1,3-Dichloropropene	0	X/Y	0	0
Ethylbenzene	X	X/Y	0	0
2-Hexanone	X	0	0	0
Methylene Chloride	X	X/Y	X/Y	0
4-Methyl-2-Pentanone	X	0	0	0
Tetrachloroethene	X	X/Y	0	0
1,1,2,2-Tetrachloroethane	0	X/Y	0	0
Toluene	X	X/Y	0	0
1,1,1-Trichloroethane	X	X/Y	X/Y	0
Trichloroethene	X	X/Y	X/Y	0
Styrene	X	0	0	0
Vinyl chloride	X	X/Y	0	0
o-,m- or p-Xylene	X	X/Y	0	0

TABLE 6-3

Chemical	Presence In Media			
	Leachate	Groundwater	Surface Water	Sediment
SEMIVOLATILES				
Phenol	X	0	0	0
* 2-Methylphenol	X	0	0	0
* 4-Methylphenol	X	0	0	0
2,4-Dimethylphenol	X	0	0	0
Benzoic Acid	X	0	0	0
Di-n-Octylphthalate	X	0	0	0
Bis (2-ethylhexyl) phthalate	X	X/Y	0	X/Y
Di-n-butylphthalate	0	0	0	X/Y
1,2-Dichlorobenzene	0	X/Y	0	0
* 1,3-Dichlorobenzene	X	X/Y	0	0
1,4-Dichlorobenzene	X	X/Y	0	0
Diethylphthalate	0	X/Y	0	0
PAHs (Noncarcinogenic)	X	X/Y	0	0
PESTICIDES/PCBs				
BHC (Hexachlorocyclohexane)	X	0	0	0
Aldrin	X	0	0	0
Heptachlor Epoxide	X	0	0	0
Endrin	X	0	0	0
Endosulfan Sulfate	X	0	0	0
Chlordane	X	0	0	0
PCBs	X	0	0	0
METALS				
Aluminum	X	0	0	0
Antimony	X	0	0	0
Arsenic	X	X/Y	0	X/NB
Barium	X	X/Y	X/NB	X/NB
Beryllium	X	0	0	0
Cadmium	X	X/Y	0	X/Y
* Calcium	X	X/NT	0	X/NT
Chromium (total)	X	X/Y	0	X/NB
Cobalt	X	X/Y	0	0
* Copper	X	X/Y	0	X/NB
* Iron	X	X/Y	0	X/NB

Chemical	Presence In Media			
	Leachate	Groundwater	Surface Water	Sediment
* Lead	X	X/Y	0	X/NB
* Magnesium	X	X/NT	0	X/NT
Manganese	X	X/Y	0	X/NB
Mercury	X	0	0	0
Nickel	X	X/Y	0	X/Y
Nitrate + Nitrite	X	X/Y	0	NA
* Potassium	X	X/NT	0	0
Selenium	X	0	0	0
Silver	X	X/Y	0	X/NB
* Sodium	X	X/Y	0	0
Thallium	X	X/Y	0	X/NB
Vanadium	X	X/Y	0	X/NB
Zinc	X	X/Y	0	X/NB
Cyanide	X	X/Y	X/Y	X/Y

Footnotes:

(1) Most organic analytes detected on-site were considered chemicals of potential concern, because they were detected above off-site background concentrations in specific media. A chemical was not considered a chemical of potential concern for either of two reasons:

a. The chemical is a nutrient of low toxicity to humans.

b. The chemical is naturally occurring and was detected at levels less than twice the average Site-specific background concentration in the particular medium.

Leachate data were used as a basis for selecting chemicals of potential concern, but a chemical was not selected as a chemical of potential concern solely because it was detected in leachate.

Refer to Section 6.2 for a more detailed discussion of the rationale used for the selection of chemicals of potential concern at the Site.

Legend

0	Not detected in medium.
X	Detected in leachate.
X/NT	Detected in medium, but not considered a chemical of potential concern, because it is a nutrient of low toxicity to humans.
X/NB	Detected in medium, but not considered a chemical of potential concern, because its concentration was within natural off-site background levels for the area.
X/Y	Detected in medium, and considered a chemical of potential concern because of its toxicity and/or its concentrations was elevated above twice the natural off-site background concentrations for the area.
*	Toxicity values (i.e., reference doses or slope factors) have not been developed by the U.S. EPA for this chemical.
NA	Parameter not analyzed.

TABLE 6-4

Chemicals Detected in Leachate and Groundwater at the WRL and Off-Site Upgradient of the WRL

<u>Volatiles</u>

Benzene	Ethylbenzene
Chloroethane	Methylene Chloride
Chloromethane	Tetrachloroethene
Chloroform	Toluene
1,1-Dichloroethane	1,1,1-Trichloroethane
1,1-Dichloroethene	Trichloroethene
1,2- cis or trans-Dichloroethene	Vinyl Chloride
1,2-Dichloropropane	Xylenes

Pesticides/PCBs	Metals
Not detected	Arsenic
	Barium
	Cadmium
	Calcium
	Chromium
	Iron
	Magnesium
	Manganese
	Zinc
	Cyanide

MWK/vlr/SGW2/JAH/JFK
[jlv-403-93y]
13160.40-MD

TABLE 6-5

Chemicals Detected in Killbuck Creek Surface Water and Sediment, Groundwater at the WRL and Upgradient of the WRL

Volatiles	Metals
Benzene	Cadmium
Chloroform	
Methylene Chloride	
1,1,1-Trichloroethane	
Trichloroethene	
cis-1,2-Dichloroethene	

TABLE 6-6
TENTATIVELY IDENTIFIED COMPOUNDS (TICs) (1)
DETECTED IN LEACHATE

COMPOUND	MAX. CONC. ug/L	MIN. CONC. ug/L	# OF (2) DETECTS	AVG. CONC. ug/L	GEOMETRIC MEAN ug/L
Semivolatiles - TICs					
Heptadecane	100.000	100.000	1	100.00	100.00
Docosane	600.000	600.000	1	600.00	600.00
Undecane	280.000	240.000	2	260.00	259.23
Dodecane, 2,7,10-Trimethyl-	570.000	570.000	1	570.00	570.00
2-Propanol, 1-[2-(2-Methoxy-1-Methylethoxy)-1-Methylethoxy]	1000.000	130.000	2	565.00	360.56
* Benzoic acid, 4-(1,1-Dimethylethyl)-	190.000	43.000	3	121.00	102.03
Decane	140.000	140.000	1	140.00	140.00
* 1,2-Benzenedicarboxylic Acid	740.000	190.000	2	465.00	374.97
Pentatriacontane	150.000	150.000	1	150.00	150.00
Iron, tricarbonyl[N-(phenyl-...	170.000	170.000	1	170.00	170.00
Octacosane	720.000	140.000	3	490.00	394.70
Octane, 2,3,6-trimethyl-	210.000	210.000	1	210.00	210.00
* Sulfur, Mol. (S8)	1700.000	140.000	3	666.67	336.43
Eicosane, 10-methyl-	170.000	150.000	2	160.00	159.69
1-Decanol, 2-ethyl-	480.000	480.000	1	480.00	480.00
Dodecane, 3-methyl-	160.000	160.000	1	160.00	160.00
Tetracontane, 3,5,24-trimethyl-	250.000	250.000	1	250.00	250.00
6,10,14-Hexadecatrien-1-ol	460.000	460.000	1	460.00	460.00
Cyclohexanone, 3,3,5-trimethyl-	93.000	60.000	2	76.50	74.70
* Camphor (ACN)	800.000	61.000	4	472.75	325.84
3-Cyclohexene-1-methanol, .alpha.,.alpha.,4-trimethyl-,(S)-	580.000	55.000	5	302.60	204.95
Cis-Terpin Hydrate	270.000	49.000	5	145.80	126.99
Benzoic acid, 4-methyl-	1200.000	120.000	6	556.67	378.41
Benzene, (1-nitropropyl)-	460.000	74.000	3	254.67	198.57
* Benzamide, n,n-diethyl-3-methyl-	260.000	79.000	5	167.40	148.54
* 2(3H)-Benzothiazolone	390.000	110.000	6	208.33	190.92
* Benzenesulfonamide, n-ethyl-4-methyl-	28.000	28.000	1	28.00	28.00

TABLE 6-6
TENTATIVELY IDENTIFIED COMPOUNDS (TICs) (1)
DETECTED IN LEACHATE

COMPOUND	MAX. CONC. ug/L	MIN. CONC. ug/L	# OF (2) DETECTS	AVG. CONC. ug/L	GEOMETRIC MEAN ug/L
Phenol,2-[1-(4-hydroxyphenyl) -1-methylethyl]-	250.000	220.000	2	235.00	234.52
2-hexen-1-ol, (Z)-	250.000	250.000	1	250.00	250.00
Pentanoic acid, 4-methyl-	1500.000	35.000	2	767.50	229.13
Hexanoic acid, 2-methyl-	240.000	240.000	1	240.00	240.00
Heptanoic acid	850.000	850.000	1	850.00	850.00
Benzeneacetic acid	2900.000	590.000	2	1745.00	1308.05
Benzenepropanoic acid	2800.000	130.000	3	1140.00	562.90
2-Naphthalenemethanol, decahydro-.alpha., .,.alpha., 4A,8-tetramethyl-,	200.000	200.000	1	200.00	200.00
Phenol, 3,4-dimethyl-	220.000	110.000	2	165.00	155.56
Bicyclo[3.1.1]heptan-2-one,	890.000	95.000	3	398.33	260.88
Benzoic acid,3,4-dimethyl-	220.000	150.000	2	185.00	181.66
* Bicyclo[2.2.1]heptan-2-one,	720.000	110.000	8	327.50	255.32
* Phenol, 2,3-dimethyl-	260.000	110.000	3	206.67	192.65
Phenol,2-(1-methylethyl)-	380.000	280.000	2	330.00	326.19
Propanedioic acid, phenyl-	100.000	100.000	1	100.00	100.00
Phenol, 3,5-dimethyl-	130.000	130.000	1	130.00	130.00
1,6-Octadien-3-ol, 3,7-dimethyl...	570.000	570.000	1	570.00	570.00
3-Cyclohexene-1-methanol,.a...	1000.000	360.000	4	647.50	605.18
Decane, 2,5,6-trimethyl-	650.000	650.000	1	650.00	650.00
3-Heptene, 7-ethoxy-	500.000	500.000	1	500.00	500.00
Cyclohexanol, 3,3,5-trimethyl-	700.000	240.000	2	470.00	409.88
Octadecane, 3-methyl-	470.000	470.000	1	470.00	470.00
Hexadecane, 3-methyl-	480.000	480.000	1	480.00	480.00
Silane, trichlorooctadecyl-	470.000	470.000	1	470.00	470.00
Decane, 3-bromo-	670.000	670.000	1	670.00	670.00
Heptadecane, 2,6-dimethyl-	170.000	130.000	4	152.50	151.76
3-Pentanol, 2,3,4-trimethyl-	240.000	240.000	1	240.00	240.00
Hexanoic acid, 3,5,5-trimethyl-	1300.000	160.000	2	730.00	456.07
Benzoic acid, 3-methyl-	660.000	660.000	1	660.00	660.00

TABLE 6-6
TENTATIVELY IDENTIFIED COMPOUNDS (TICs) (1)
DETECTED IN LEACHATE

COMPOUND	MAX. CONC. ug/L	MIN. CONC. ug/L	# OF (2) DETECTS	AVG. CONC. ug/L	GEOMETRIC MEAN ug/L
Butanoic acid, 2-methylcyclo...	250.000	250.000	1	250.00	250.00
Benzenebutanoic acid, 2,5-di...	210.000	210.000	1	210.00	210.00
1,4-Dioxane, 2-ethyl-5-methyl-	440.000	440.000	1	440.00	440.00
13-Tetradecynoic acid, methy...	250.000	250.000	1	250.00	250.00
Cyclohexane, [2-[(2-ethylhex...	450.000	450.000	1	450.00	450.00
4-Octadecenal	270.000	270.000	1	270.00	270.00
Ethanone, 1-(1-cyclohexen-1-...	560.000	560.000	1	560.00	560.00
1(2H)-Naphthalenone, octahyd...	220.000	220.000	1	220.00	220.00
Butanoic acid, 2-ethyl-,1,2...	690.000	690.000	1	690.00	690.00
Butanoic acid, 3,3-dimethyl-	400.000	400.000	1	400.00	400.00
2,7-Nonadien-5-one,4,6-dime...	1800.000	1800.000	1	1800.00	1800.00
Hexane, 2-(hexyloxy)-5-methy...	540.000	540.000	1	540.00	540.00
.beta.-d-glucopyranoside,me...	1100.000	1100.000	1	1100.00	1100.00
Cyclohexanol, 4-(1-methyleth...	500.000	500.000	1	500.00	500.00
4-Heptanol, 3,4-dimethyl-	570.000	570.000	1	570.00	570.00
14-Pentadecynoic acid, methy...	180.000	180.000	1	180.00	180.00
3-Benzofurancarboxylic acid,...	230.000	230.000	1	230.00	230.00
1-Heptanol, 2-propyl-	180.000	180.000	1	180.00	180.00
Phenol, 3-(1-methylethyl)-	76.000	76.000	1	76.00	76.00
Bicyclo[2.2.1]heptane,2,5-di...	270.000	77.000	2	173.50	144.19
Cyclohexane, (1,1-dimethylpr...	80.000	80.000	1	80.00	80.00
Bycyclo[3.1.1]heptane-2-carb...	100.000	100.000	1	100.00	100.00
Benzene, 2-methoxy-1,3,4-tri...	77.000	77.000	1	77.00	77.00
3-Heptyne, 5,5-diethyl-...	130.000	130.000	1	130.00	130.00

TABLE 6-6
TENTATIVELY IDENTIFIED COMPOUNDS (TICs) (1)
DETECTED IN LEACHATE

COMPOUND	MAX. CONC. ug/L	MIN. CONC. ug/L	# OF (2) DETECTS	AVG. CONC. ug/L	GEOMETRIC MEAN ug/L
Methanone, [4-(1,1-dimethyle...	100.000	100.000	1	100.00	100.00
Nonadecane, 2,3-dimethyl-	87.000	87.000	1	87.00	87.00
Benzenemethanol,.alpha.,.al...	180.000	180.000	1	180.00	180.00
Phenol, 3-propyl-	200.000	200.000	1	200.00	200.00
.Alpha.-santalol	390.000	390.000	1	390.00	390.00
Decane, 4-methyl-	170.000	170.000	1	170.00	170.00
6-Octen-1-ol, 3,7-dimethyl-	150.000	150.000	1	150.00	150.00
4-Nonenal, (E)-	160.000	160.000	1	160.00	160.00
Undecane, 5-ethyl-	140.000	140.000	1	140.00	140.00
Oxirane, tetradecyl-	460.000	460.000	1	460.00	460.00
Propanedioic acid, dimethyl-	26.000	26.000	1	26.00	26.00
Cyclohexanol, 1,1'-dioxybis-	49.000	49.000	1	49.00	49.00
Butanoic acid	86.000	86.000	1	86.00	86.00
Butanoic acid, 2-methyl-	48.000	48.000	1	48.00	48.00
Hexanoic acid, (DOT)	94.000	94.000	1	94.00	94.00
2-Pyrrolidininone, 1-methyl-	71.000	71.000	1	71.00	71.00
2-Propanol, 1-[2-2(2-methoxy-...	270.000	270.000	1	270.00	270.00
Benzeneacetic acid, .alpha.-	140.000	140.000	1	140.00	140.00
Cyclopentasiloxane, decamety...	240.000	240.000	1	240.00	240.00
Hexadecane, 7-methyl-	110.000	110.000	1	110.00	110.00
Heptadecane, 2-methyl-	100.000	100.000	1	100.00	100.00
4-Hexenoic acid, 3-methyl-2,...	150.000	150.000	1	150.00	150.00
Cholestane, 4,5-epoxy-, (4.A...	160.000	160.000	1	160.00	160.00
Cholestan-3-one, 4,4-dimethy...	160.000	160.000	1	160.00	160.00
Volatiles - TICs					
* Silanol, trimethyl	74.000	19.000	11	48.45	42.21
4-Penten-2-ol	51.000	44.000	2	47.50	47.37
* Furan, tetrahydro-	230.000	11.000	12	80.50	51.44
2-Butanol, 3-methyl-	160.000	140.000	2	150.00	149.67

TABLE 6-6
TENTATIVELY IDENTIFIED COMPOUNDS (TICs) (1)
DETECTED IN LEACHATE

COMPOUND	MAX. CONC. ug/L	MIN. CONC. ug/L	# OF (2) DETECTS	AVG. CONC. ug/L	GEOMETRIC MEAN ug/L
2-Butanone, 3-methyl-	110.000	6.900	5	48.78	26.80
2-Pentanol, 4-methyl-	25.000	24.000	2	24.50	24.49
* 3-Pentanone, 2,4-dimethyl-	28.000	5.600	9	17.99	15.96
2-Hexanone, 5-methyl-	30.000	6.800	5	18.72	16.16
Cineole (van)	43.000	6.800	7	19.77	16.01
* Bicyclo[2.2.1]heptan-2-one, 1,7,7-trimethyl-,(+-)-	140.000	9.300	12	47.44	36.55
3-Carene	27.000	10.000	3	18.00	16.62
* Ethyl ether	32.000	7.700	3	16.13	12.89
2-Butanol, 2-methyl-	15.000	15.000	1	15.00	15.00
.Alpha.-pinene (ACN)	42.000	42.000	1	42.00	42.00
7-Oxabicyclo[2.2.1]heptane-1-methyl-4-(1-methylethyl)-	14.000	14.000	1	14.00	14.00
4-Carene, (1S,3R,6R)-(-)-	65.000	40.000	2	52.50	50.99
Bicyclo[3.1.1]heptane, 6,6-d...	49.000	49.000	1	49.00	49.00
1,3-Oxathiolane	19.000	18.000	2	18.50	18.49
2-Pentanone, 3-methyl-	47.000	8.900	4	29.48	23.56
2,3-Hexanedione	27.000	27.000	1	27.00	27.00
1,1'-Bicyclopentyl	18.000	18.000	1	18.00	18.00
1,6-Octadien-3-ol, 3,7-dimet...	28.000	17.000	2	22.50	21.82
2-Butanol	47.000	47.000	1	47.00	47.00
2-Pentanone	54.000	54.000	1	54.00	54.00
Camphor (ACN)	19.000	19.000	1	19.00	19.00

Notes:

* Compounds tentatively identified in both leachate and groundwater.

1. This summary includes the five rounds of leachate data. The number of detects reflects the total number of samples in which the TIC was detected. The number of locations where a TIC was detected may be less than the total number of detects.

2. Twelve samples were analyzed for semivolatiles and 25 samples for volatiles.

CAW/caw/TJD/MWK/JFK

TABLE 6-7
TENTATIVELY IDENTIFIED COMPOUNDS (TICs) (1)
DETECTED IN GROUNDWATER

COMPOUND	MAX. CONC. ug/L	MIN. CONC. ug/L	# OF (2) DETECTS	AVG. CONC. ug/L	GEOMETRIC MEAN ug/L
Semivolatiles - TICs					
Hexadecanoic Acid	10.000	10.000	1	10.00	10.00
* Benzoic acid, 4-(1,1-Dimethylethyl)-	17.000	10.000	3	12.67	12.32
* 1,2-Benzenedicarboxylic Acid	18.000	9.300	4	12.95	12.43
* Sulfur, Mol. (S8)	650.000	8.200	9	117.13	52.95
* Camphor (ACN)	14.000	14.000	1	14.00	14.00
* Benzamide, n,n-diethyl-3-methyl-	10.000	9.400	3	9.80	9.80
* 2(3H)-Benzothiazolone	30.000	11.000	6	19.00	18.01
* Benzenesulfonamide, n-ethyl-4-methyl-	14.000	11.000	2	12.50	12.41
* Bicyclo[2.2.1]heptan-2-one,	26.000	26.000	1	26.00	26.00
* Phenol, 2,3-dimethyl-	18.000	9.300	2	13.65	12.94
Phenol, 4-(1-methylethyl)-	9.300	9.300	1	9.30	9.30
Benzamide, n-(1,1-dimethylethyl)-4-methyl-	20.000	20.000	1	20.00	20.00
Hexanedioic acid, bis (2-ethyl...)	13.000	8.400	2	10.70	10.45
Benzenesulfonamide, n-butyl-...	10.000	9.500	2	9.75	9.75
3,6-Dioxa-2,4,5,7-Tetrasilao...	17.000	17.000	1	17.00	17.00
Ethane, 1,1'-Oxybis[2-ethoxy...	8.400	8.400	1	8.40	8.40
1,3-Pentanediol, 2,2,4-trime...	15.000	15.000	1	15.00	15.00
1-Propanol, 2-(2-methoxy-1-m...	17.000	17.000	1	17.00	17.00
1-Hexene, 3,4,5-trimethyl-	9.800	9.800	1	9.80	9.80
Benzenesulfonamide, n-ethyl-	31.000	31.000	1	31.00	31.00
Pentanamide, 4-methyl-	30.000	30.000	1	30.00	30.00
Benzoic acid, 4-(1,1-dimethyl)-	14.000	14.000	1	14.00	14.00
9-Octadecenamide, (Z)-	13.000	13.000	1	13.00	13.00

TABLE 6-7
TENTATIVELY IDENTIFIED COMPOUNDS (TICs) (1)
DETECTED IN GROUNDWATER

COMPOUND	MAX. CONC. ug/L	MIN. CONC. ug/L	# OF (2) DETECTS	AVG. CONC. ug/L	GEOMETRIC MEAN ug/L
Volatiles - TICs					
* Silanol, trimethyl	13.000	5.500	2	9.25	8.46
Benzene, 1,4-dichloro-	5.500	5.500	1	5.50	5.50
* Furan, tetrahydro-	23.000	9.800	2	16.40	15.01
* 3-Pentanone, 2,4-dimethyl-	5.500	5.500	1	5.50	5.50
* Bicyclo[2.2.1]heptan-2-one, 1,7,7-trimethyl-,(+-)-	9.800	9.800	1	9.80	9.80
* Ethyl ether	130.000	5.600	12	34.72	22.69
Methane, chlorofluoro-	52.000	5.000	5	22.40	15.52
Methane, dichlorofluoro-	44.000	6.300	6	17.50	12.67
Methane, chlorodifluoro-	16.000	16.000	1	16.00	16.00
Ethane, 1,1'-thiobis	8.500	8.500	1	8.50	8.50
Ethane, 1,1'-[methylenebis(o...	8.900	8.900	1	8.90	8.90
Methane, thiobis-	7.200	7.200	1	7.20	7.20
Benzene, 1,2-dichloro-	16.000	16.000	1	16.00	16.00

Notes:

* Compounds tentatively identified in both leachate and groundwater.

1. This summary includes the four rounds of groundwater data. The number of detects reflects the total number of samples in which the TIC was detected. The number of locations where a TIC was detected may be less than the total number of detects.

2. Twenty-four samples were analyzed for semivolatiles and 64 for volatiles.

CAW/caw/MWK/TJD/KJD/JFK

TABLE 6-8
TENTATIVELY IDENTIFIED COMPOUNDS (TICs) (1)
DETECTED IN CREEK SEDIMENT

COMPOUND	MAX. CONC. ug/kg	MIN. CONC. ug/kg	# OF (2) DETECTS	AVG. CONC. ug/kg	GEOMETRIC MEAN ug/kg
Semivolatiles - TICs					
Octacosane	270.000	170.000	2	220.00	214.24
Eicosane, 10-methyl-	160.000	160.000	1	160.00	160.00
Unknown Phthalate	4400.000	4400.000	1	4400.00	4400.00
Hexanedioic acid, dioctylester	230.000	230.000	1	230.00	230.00
Phosphoric acid, 2-ethylhexyl diphenyl ester	1100.000	1100.000	1	1100.00	1100.00

Notes:

1. The number of detects reflects the total number of samples in which the TIC was detected. The number of locations where a TIC was detected may be less than the total number of detects.

2. Five samples were analyzed.

CAW/caw/MWK/TJD/KJD/JFK

TABLE 6-9

Physical/Chemical Properties of Target Compound List Organics
Chemicals Detected (a)

	Molecular Weight (g/mole)	Water Solubility (mg/L)	Vapor Pressure (mm Hg)	Henry's Constant (atm-m^3/mole)	Organic Carbon Partition Coefficient (ml/g)	Octanol Water Partition Coefficient (log10)
Volatiles						
Chloromethane	--	--	--	--	--	--
Vinyl Chloride	63	2.67E+03	2.66E+03	8.19E-02	5.7E+01	1.38
Chloroethane	--	--	--	--	--	--
Methylene Chloride	84.93	2.00E+04	3.49E+02	--	--	--
Acetone	58	1.00E+06	2.70E+02	2.06E-05	2.2E+00	-0.24
1,1-Dichloroethene	96.95	--	5.00E+02	--	--	--
1,1-Dichloroethane	99	5.50E+03	1.82E+02	4.31E-03	3.0E+01	1.79
1,2-Dichloroethene (total)	97	6.30E+03	3.24E+02	6.56E-03	5.9E+01	0.48
Chloroform	119	8.20E+03	1.51E+03	2.87E-03	3.1E+01	1.97
1,2-Dichlorethane	99	8.52E+03	6.40E+01	9.78E-04	1.4E+01	1.48
2-Butanone	72.1	3.53E+02	7.75E+01	--	--	--
1,1,1-Trichloroethane	133	1.50E+03	1.23E+02	1.44E-02	1.5E+02	2.50
Carbon Tetrachloride	154	7.57E+02	9.00E+01	2.41E-02	1.1E+02	2.64
Bromodichlormethane	163.8	--	--	--	--	--
1,2-Dichloropropane	113	2.70E+03	4.20E+01	2.31E-03	5.1E+01	2.00
cis-1,2-Dichloroethene	96.95	8.00E+02	2.00E+02	--	--	--
Trichloroethene	131	1.10E+03	5.79E+01	9.10E-03	1.3E+02	2.38
Dibromochloromethane	208.3	--	--	--	--	--
1,1,2-Trichloroethane	133	4.50E+03	3.00E+01	1.17E-03	5.6E+01	2.47
Benzene	78	1.75E+03	9.52E+01	5.59E-03	8.3E+01	2.12
Trans-1,3-Dichloropropene	111	2.,80E+03	2.50E+01	1.30E-03	4.8E+01	2.00
Bromoform	252.77	3.19E+03	5.60E+00	--	--	--
4-Methyl-2-Pentanone	100.2	1.70E+04	6.00E+00	--	--	--
2-Hexanone	100.2	3.50E+04	2.00E+00	--	--	--
Tetrachloroethene	166	1.50E+02	1.78E+01	2.59E-02	3.6E+02	2.60
1,1,2,2-Tetrachloroethane	168	2.90E+03	5.00E+00	3.81E-04	1.18E+02	2.39
Toluene	92	5.35E+02	2.81E+01	6.37E-03	3.0E+02	2.73
Chlorobenzene	113	4.66E+02	1.17E+01	3.72E-03	3.3E+02	2.84
Ethylbenzene	106	1.52E+02	7.00E+00	6.43E-03	1.1E+03	3.15
Styrene	104	--	--	--	--	--
Xylene (total)	106	1.98E+02	1.00E+01	7.04E-03	2.4E+02	3.26
trans-1,2,-Dichloroethene	96.95	6.00E+02	2.00E+02	--	--	--
Semivolatiles						
1,4-Dichlorobenzene	147	7.90E+01	1.18E+00	2.89E-03	1.7E+03	3.60
1,3-Dichlorobenzene	147	1.23E+02	2.28E+00	3.59E-03	1.7E+03	3.60
1,2-Dichlorobenzene	147	1.00E+02	1.00E+00	1.93E-03	1.7E+03	3.60
2-Methylphenol	108.13	3.10E+04	2.40E-01	--	--	--
2,4-Dimethylphenol	122.16	--	--	--	--	--
Acenaphthene	154	3.42E+00	1.55E-03	9.20E-05	4.6E+03	4.00
Dibenzofuran	--	--	--	--	--	--
Di-n-butylphthalate	278	1.30E+01	1.00E-05	2.82E-07	1.7E+05	5.60
Diethylphthalate	222	8.96E-02	3.50E-03	1.14E-06	1.4E+12	2.50
Fluoranthene	202	2.06E-01	5.00E-06	6.46E-06	3.8E+04	4.90
Pyrene	202	1.32E-01	2.50E-06	5.04E-06	3.8E+04	4.88
Benzo(a)anthracene	228	5.70E-03	2.20E-08	1.16E-06	1.4E+06	5.60
Chrysene	228	1.80E-03	6.30E-09	1.05E-06	2.0E+05	6.06
Bis(2-Ethylhexyl) phthalate	391	--	--	--	5.8E+03(1)	--
Benzo(b)fluoranthene	252	1.40E-02	5.00E-07	1.19E-05	5.5E+05	6.06
Benzo(a)pyrene	252	1.20E-03	5.60E-09	1.55E-06	5.5E+06	6.06
Benzo(k)fluoranthene	252	4.30E-03	5.10E-07	3.94E-05	5.5E+05	6.06
Pesticides/PCBs						
PCBs	328	3.10E-02	7.70E-05	1.07E-03	5.3E+05	6.04

TABLE 6-9
(Continued)

Notes

(1) The Koc for this phthalate was provided by the Illinois EPA (August 3, 1990).

Reference

(a) All values were obtained from the U.S. EPA Superfund Public Health Evaluation Manual (SPHEM), 1986 unless otherwise referenced. (--) indicates property not identified.

Definitions

1. Water Solubility is the maximum concentration of a chemical that dissolves in pure water at a specific temperature and pH. Values are given for a neutral pH and a temperature range of 20 to 40° C. The rate at which a chemical is leached from a waste by infiltrating precipitation is a function of its solubility in water. The more soluble compounds are expected to be leached more readily and rapidly than less soluble chemicals. The water solubilities presented in the literature indicate that the volatile organic chemicals are usually several orders of magnitude more water soluble than the base/neutral organic compounds (e.g., PAHs).

2. Vapor pressure provides an indication of the rate at which a chemical in its pure state volatilizes. Values are given for a temperature range of 20 to 30° C. It is of primary significance where environmental interfaces such as surface soil/air and surface water/air occur. Volatilization is not as important when evaluating groundwater and subsurface soils. Chemicals with higher vapor pressures are expected to enter the atmosphere more readily than chemicals with lower vapor pressures. Vapor pressures for monocyclic aromatics (toluene) and chlorinated aliphatics (TCE) are generally many times higher than vapor pressure for phthalate esters (bis(2-ethylhexyl)phthalate), PAHs, and pesticides.

3. Henry's Law Constant is important in evaluating air exposure pathways. Values for Henry's Law Constant (H) were calculated using the following equation and the values previously recorded for solubility, vapor pressure, and molecular weight:

$$H(\text{atm-m}^3/\text{mole}) = \frac{\text{Vapor Pressure (atm)} \times \text{Mole Weight (g/mole)}}{\text{Water Solubility (g/m}^3\text{)}}$$

4. Organic Carbon Partition Coefficient (Koc) is a measure of the tendency for organics to be adsorbed by soil and sediment and is expressed as:

$$\text{Koc} = \frac{\text{mg chemical adsorbed/kg organic carbon}}{\text{mg chemical dissolved/liter of solution}}$$

The Koc is chemical specific and is largely independent of soil properties. In general, the compound's Koc is inversely related to its environmental mobility.

5. The octanol/water partition coefficient is used to estimate bioconcentration factors in aquatic organisms. A linear relationship between the octanol/water partition coefficient and the uptake of chemicals by fatty tissues of animal and human receptors (bioconcentration factor) has been determined. It is also useful in estimating the sorption and desorption of compounds by organic soils, where experimental values are not available.

MWK/vlr/TJD/JFK
[jlv-403-93c]
13160.40-MD

TABLE 6-10

Toxicity Profiles for Selected Chemicals of Potential Concern(1)

Chemical	Reference Doses			Slope Factors		
	Oral			Oral		
	Toxic Effect	Species	Uncertainty Factor	Cancer Site	Species	Cancer Classification(2)
VOLATILES						
1,2-Dichloroethene (cis)	Decreased hematocrit and hemoglobin	Rat	3000	--	--	--
1,2-Dichloroethene (trans)	Increased serum enzyme levels	Mouse	100	--	--	--
Vinyl Chloride	NA	NA	NA	Lung	Rat	A
METALS						
Arsenic	Keratosis and hyper-pigmentation	Human	1	Skin	Human	A
Barium	Increased blood pressure	Rat	NA	--	--	D
Cobalt	NA	NA	NA	--	--	--
Managanese	Central nervous system effects	Rat	100	--	--	D
Thallium	Increased serum enzyme levels, loss of hair	Rat	3000	--	--	D
Zinc	Anemia	Human	10	--	--	--

Notes

(1) Data presented were summarized from the Integrated Risk Information System (IRIS) and the fourth quarter (September 1990) Health Effects Assessment Summary Tables (HEAST). This table summarizes the health effects for those chemicals which contributed substantially (i.e., >5%) to the total health risk due to groundwater exposure at the WRL. Health risks for other media (i.e., surface water and sediment) were below U.S. EPA risk goals; therefore, exposure to chemicals in these media were not considered a health concern and was not included in this table. Chronic lengths of exposure were assumed in the BRA; therefore, the effects due to chronic exposure have been summarized. Data on dermal exposure are not available, but it was assumed that upon absorption, the chemical would exhibit effects similar to the oral route of exposure.

(2) The code represents the U.S. EPA weight-of-evidence classification system for carcinogenicity for the particular chemical. The following is a description of the classification by group.

TABLE 6-10

Group	Description
A	Human carcinogen
B1 or B2	Probable human carcinogen
	B1 indicates that limited human data are available.
	B2 indicates sufficient evidence in animals and inadequate or no evidence in humans.
C	Possible human carcinogen
D	Not classifiable as to human carcinogenicity
E	Evidence of noncarcinogenicity for humans

Legend

-- = not applicable

NA = not available

TABLE 6-11

Critical Toxicity Values(1)

CHEMICAL	CHRONIC REFERENCE DOSES (mg/kg/day)(2)		SLOPE FACTORS (mg/kg/day)-1	
	ORAL	DERMAL(3)	ORAL	DERMAL(3)
VOLATILES				
Acetone	1.0e-01	8.0e-02	ND	ND
Benzene	ND	ND	2.9e-02	5.8e-02
Bromodichloromethane	2.0e-02	1.0e-02	1.3e-01	2.6e-01
Bromoform	2.0e-02	1.0e-02	7.9e-03	1.6e-02
Carbon Tetrachloride	7.0e-04	3.5e-04	1.3e-01	2.6e-01
Chlorobenzene	2.0e-02	1.0e-02	ND	ND
Chloroethane	ND	ND	ND	ND
Chloroform	1.0e-02	1.0e-02	6.1e-03	6.1e-03
Chloromethane	ND	ND	1.3e-02	2.6e-02
Dibromochloromethane	2.0e-02	1.0e-02	8.4e-02	1.7e-01
1,1-Dichloroethane	1.0e-01	5.0e-02	ND	ND
1,2-Dichloroethane	ND	ND	9.1e-02	9.1e-02
1,1-Dichloroethene	9.0e-03	4.5e-03	6.0e-01	1.2e+00
1,2-Total Dichloroethene(4)	1.0e-02	1.0e-02	ND	ND
1,2-Dichloropropane	ND	ND	6.8e-02	1.4e-01
1,3-Dichloropropene	3.0e-04	1.5e-04	1.8e-01	3.6e-01
Ethylbenzene	1.0e-01	5.0e-02	ND	ND
Methylene Chloride	6.0e-02	4.2e-02	7.5e-03	1.1e-02
Tetrachloroethene	1.0e-02	5.0e-03	5.1e-02	1.0e-01
1,1,2,2-Tetrachloroethane	ND	ND	2.0e-01	2.0e-01
Toluene	2.0e-01	2.0e-01	ND	ND
1,1,1-Trichloroethane	9.0e-02	9.0e-02	ND	ND
Trichloroethene	ND	ND	1.1e-02	1.1e-02
Vinyl chloride	ND	ND	2.3e+00	4.6e+00
Xylene (mixed)	2.0e+00	2.0e+00	ND	ND
SEMIVOLATILES				
Diethyl Phthalate	8.0e-01	4.0e-01	ND	ND
1,2-Dichlorobenzene	9.0e-02	4.5e-02	ND	ND
1,4-Dichlorobenzene	ND	ND	2.4e-02	4.8e-02
Bis(2-ethylhexyl)Phthalate	2.0e-02	1.0e-02	1.4e-02	2.8e-02
PAHS (Noncarcinogenic) (5)	4.0e-03	2.0e-03	ND	ND
METALS				
Arsenic (6)	1.0e-03	1.0e-03	1.8e+00	1.8e+00
Barium	5.0e-02	2.5e-03	ND	ND
Cadmium	5.0e-04	1.0e-05	ND	ND
Chromium (total)(7)	1.0e+00	3.0e-02	ND	ND
Cobalt (8)	1.0e-05	5.0e-07	ND	ND
Manganese	1.0e-01	4.0e-03	ND	ND

CHEMICAL	CHRONIC REFERENCE DOSES (mg/kg/day)(2)		SLOPE FACTORS (mg/kg/day)-1	
	ORAL	DERMAL(3)	ORAL	DERMAL(3)
METALS (cont.)				
Nickel	2.0e-02	1.0e-03	ND	ND
Nitrates	1.0e-01	5.0e-03	ND	ND
Silver	3.0e-03	1.5e-04	ND	ND
Thallium	7.0e-05	3.5e-06	ND	ND
Vanadium	7.0e-03	3.5e-04	ND	ND
Zinc	2.0e-01	5.0e-02	ND	ND
Cyanide	2.0e-02	1.4e-02	ND	ND

Notes

(1) The critical toxicity factors were obtained from the Integrated Risk Information System (IRIS) or the fourth quarter Health Effects Assessment Summary Tables (HEAST), September 1990.

(2) Chronic reference doses were used if exposure was considered to be chronic (i.e., ≥7 years or 10 percent of the average human lifespan). Subchronic reference doses were used for shorter lengths of chemical exposure.

(3) The dermal toxicity factors were estimated by adjusting the oral toxicity factor (administered doses) using the percent bioavailability of the compound via the oral route (see Table 6-12). Therefore, the resultant dermal toxicity factors are based on the absorbed dose of the particular chemical. The following equations were used to adjust the oral toxicity factor to a dermal toxicity factor.

Reference Doses (RFD)

RFD oral x % oral bioavailability (i.e., 1 = 100%) = RFD dermal

Slope Factors (SF)

SF oral ÷ % oral bioavailability (i.e., 1 = 100%) = SF dermal

(4) The critical toxicity values for cis 1,2-dichloroethene were used to calculate the health risks for 1,2-dichloroethene because this isomer is the more toxic of the two isomers. Also, the predominant isomer detected in groundwater at the WRL was cis 1,2-dichloroethene.

(5) The Naphthalene reference dose is used to estimate the toxicity of noncarcinogenic PAHs.

(6) The reference dose was derived from the unit risk value provided in HEAST.

(7) Chromium risks were estimated assuming the valence state of the element is +3 in each medium. Based on the probable groundwater and sediment conditions near the Site (reducing or mild oxidizing conditions), chromium would exist in the +3 state.

(8) The chronic reference dose is an interim value provided by the Environmental Criteria and Assessment Office (ECAO) of the U.S. EPA.

Legend

ND = critical toxicity factor has not been determined by the U.S. EPA.

MWK/kml/TJD/JFK [jlv-403-93h] 13160.40-MD

TABLE 6-12

Percent Oral Bioavailability of Contaminants

CHEMICAL	ORAL BIOAVAILABILITY	BASIS FOR ESTIMATE (1)
VOLATILES		
Acetone	0.80	H
Benzene	0.50	E
Bromodichloromethane	0.50	E
Bromoform	0.50	E
Carbon Tetrachloride	0.50	B
Chlorobenzene	0.50	E
Chloroethane	0.50	E
Chloroform	1.00	H
Chloromethane	0.50	E
Dibromochloromethane	0.50	E
1,1-Dichloroethane	0.50	C
1,2-Dichloroethane	1.00	A
1,1-Dichloroethene	0.50	B
1,2-TotalDichloroethene	1.00	C
1,2-Dichloropropane	0.50	E
1,3-Dichloropropene	0.50	E
Ethylbenzene	0.50	B
Methylene Chloride	0.70	F
Tetrachloroethene	0.50	B
1,1,2,2-Tetrachloroethane	1.00	F
Toluene	1.00	H
1,1,1-Trichloroethane	1.00	H
Trichloroethene	1.00	H
Vinyl chloride	0.50	A
Xylene	1.00	H
SEMIVOLATILES		
Diethylphthalate	0.50	E
Bis(2-ethylhexyl)Phthalate	0.50	E
Di-n-butylphthalate	0.50	E
1,2-Dichlorobenzene	0.50	E
1,4-Dichlorobenzene	0.50	E
PAHS (Noncarcinogenic)	0.50	G
METALS		
Arsenic	1.00	H
Barium	0.05	E
Cadmium	0.02	H
Chromium (total)(2)	0.03	H
Cobalt	0.05	E
Manganese	0.04	H
Nickel	0.05	H
Nitrates	0.05	E
Silver	0.05	E
Thallium	0.05	E
Vanadium	0.05	E
Zinc	0.25	H
Cyanide	0.70	H

NOTES

(1) The following are the basis used for estimating the oral bioavailability of each contaminant. Values used were either cited or estimated based on data provided in the Health Effects Assessment (HEA) (U.S. EPA, 1984) for the contaminants.

A. Based on data that showed half of the chemical dose was absorbed in a very short time (i.e., within minutes).

B. Reported to be readily absorbed but no specific value cited, therefore assumed 50 percent absorption.

C. Estimated value in HEA based on another similar compound.

D. Reported in HEA as less than a specific percent oral bioavailability.

E. No information on oral bioavailability readily available. Assumed 50 percent (organics) and 5 percent (inorganics) based on data for similar chemicals where data are available.

F. Based on inhalation absorption data.

G. Based on the bioavailability of benzo(a)pyrene.

H. Based on cited value in HEA.

(2) Chromium risks are estimated assuming the valence state of the element is +3 in each medium. Based on the probable groundwater and sediment conditions near the Site (reducing or mild oxidizing conditions), chromium would exist in the +3 state.

MWK/jkk/BJC/JFK
[jlv-403-93i]
13160.40-MD

TABLE 6-13

Summary of Exposure Pathway Evaluation

Potentially Exposed Population	Exposure Route, Medium, and Exposure Point	Pathway Selected for Risk Evaluation	Rationale[1]
		Current Land Use	
Nearby Residents	Ingestion, inhalation, dermal exposure to groundwater from local wells downgradient of the Site.	No	Downgradient private well has not been affected by Site.
Nearby Residents	Dermal contact and incidental ingestion of waste/leachate from the facility.	No	Access to landfill is restricted by fencing.
Nearby Residents	Dermal contact and incidental ingestion of surface water and sediment in Killbuck Creek.	Yes	Stream sediment and surface water may have been affected by the Site. Health risks have been qualitatively addressed.
Nearby Residents	Inhalation exposure to fugitive chemical emissions released to the air.	Yes	The facility has a gas collection and destruction system in place to minimize releases of chemicals to the atmosphere. Risks have been qualitatively addressed.
Nearby Residents	Ingestion of food.	No	Food crops are not grown in the area downgradient of the WRL.
Aquatic and Terrestrial Species	Exposure to surface water and sediment in Killbuck Creek.	Yes	Risks due to surface water and sediment exposure to flora and fauna have been addressed qualitatively.

TABLE 6-13
(continued)

Potentially Exposed Population	Exposure Route, Medium, and Exposure Point	Pathway Selected for Risk Evaluation	Rationale[1]
	------------------------------ Future Land Use	------------------------------	
Nearby Residents	Ingestion and dermal exposure to groundwater from local wells down-gradient of the Site.	Yes	Downgradient private wells may be affected in the future by the Site.
Nearby Residents	Dermal contact and incidental ingestion of waste from the facility.	No	Not applicable.
Nearby Residents	Dermal contact and incidental ingestion of surface water and sediment in Killbuck Creek.	Yes	Health risks were quantitatively addressed under current Site conditions. Conditions are not expected to change appreciably in the future.
Nearby Residents	Inhalation exposure to fugitive chemical emissions released to the air.	Yes	The facility has a gas collection and destruction system in place to minimize releases of chemicals to the atmosphere. The risk was qualitatively addressed.
Nearby Residents	Ingestion of food.	No	Food crops are not grown in the area downgradient of the WRL.
Aquatic and Terrestrial Species	Exposure to surface water and sediment in Killbuck Creek.	Yes	Risks due to surface water and sediment exposure to flora and fauna have been addressed qualitatively.

(1) The rationale provided in this table is not all-inclusive. The reader is referred to the main body of the text (i.e., Section 6.2) for a full rationale.

TABLE 6-14

Exposure Point Concentrations for Chemicals by Media(1)

Chemical	Future Site Conditions: Groundwater (mg/L)	Current Site Conditions: Surface Water (mg/L)	Current Site Conditions: Sediment (mg/kg)
VOLATILES			
Acetone	4.8e-03	ND	ND
Benzene	3.5e-03	1.7e-03	ND
Bromodichloromethane	1.1e-04	ND	ND
Bromoform	2.2e-04	ND	ND
Carbon Tetrachloride	4.9e-04	ND	ND
Chlorobenzene	1.8e-03	ND	ND
Chloroethane	3.8e-02	ND	ND
Chloroform	3.0e-03	2.9e-04	2.0e-03
Chloromethane	2.3e-03	ND	ND
Dibromochloromethane	1.5e-04	ND	ND
1,1-Dichloroethane	6.9e-02	ND	ND
1,2-Dichloroethane	1.3e-03	ND	ND
1,1-Dichloroethene	2.6e-04	ND	ND
1,2-Total Dichloroethene	2.4e-01	6.7e-04	ND
1,2-Dichloropropane	7.3e-03	ND	ND
1,3-Dichloropropene	4.3e-04	ND	ND
Ethylbenzene	1.8e-03	ND	ND
Methylene Chloride	2.6e-03	1.9e-02	ND
Tetrachloroethene	2.3e-02	ND	ND
1,1,2,2-Tetrachloroethane	3.5e-03	ND	ND
Toluene	2.1e-04	ND	ND
1,1,1-Trichloroethane	5.9e-03	7.0e-04	ND
Trichloroethene	5.3e-02	1.0e-03	ND
Vinyl chloride	1.4e-02	ND	ND
Xylene	2.1e-03	ND	ND
SEMIVOLATILES			
Diethylphthalate	2.3e-03	ND	ND
Bis(2-ethylhexyl)Phthalate	4.6e-03	ND	6.3e-02
Di-n-butylphthalate	ND	ND	4.4e+00
1,2-Dichlorobenzene	1.3e-03	ND	ND
1,4-Dichlorobenzene	1.5e-02	ND	ND
PAHS (Noncarcinogenic)	8.3e-05	ND	ND
METALS			
Arsenic	8.4e-03	ND	4.0e+00
Barium	6.6e-01	8.9e-02	8.2e+01
Cadmium	2.3e-03	ND	1.0e+00
Chromium (total)	7.6e-04	ND	8.8e+00
Cobalt	3.1e-02	ND	ND
Manganese	1.3e+00	ND	2.4e+02
Nickel	3.8e-02	ND	9.7e+00
Nitrates	1.2e-02	ND	NA
Silver	2.1e-03	ND	1.0e-01
Thallium	2.8e-03	ND	4.0e-01
Vanadium	2.9e-02	ND	2.9e-02
Zinc	6.3e+00	ND	5.6e+00
Cyanide	9.4e-02	1.7e-02	2.7e-01

TABLE 6-14
(continued)

Notes

(1) Values represent the lesser of either the 95 percent upper-bound confidence limit of the arithmetic mean or maximum concentration detected in a particular medium.

Legend

ND = chemical not detected in medium
NA = not applicable

TABLE 6-15

Average Daily Absorbed Doses of Contaminants in Surface Water (1)

CHEMICAL	NONCARCINOGENIC EFFECTS	CARCINOGENIC EFFECTS
	DERMAL CONTACT	DERMAL CONTACT
VOLATILES		
Benzene	4.6e-09	1.0e-09
1,2-Total Dichloroethene	1.8e-09	3.8e-10
Chloroform	7.7e-10	1.7e-10
Methylene Chloride	5.1e-08	1.1e-08
1,1,1-Trichloroethane	1.9e-09	4.0e-10
Trichloroethene	2.7e-09	5.7e-10
METALS		
Barium	2.4e-07	5.1e-08
Cyanide	4.5e-08	9.7e-09

Notes

1. a. The average daily absorbed dose for each chemical is expressed in units of mg/kg/day.

   b. For noncarcinogenic effects, the absorbed dose (AD) is averaged over the exposure period, while for carcinogenic effects, they are averaged over a lifetime (i.e., 70 years). Therefore, the differences between the AD for noncarcinogenic vs. carcinogenic effects are due to the different methods of time weighing which are used to estimate the value.

MWK/kml/BJC/KJD/JFK
[jlv-403-93s]
13160.40-MD

TABLE 6-16

Chronic Daily Intakes and Absorbed Doses of Chemicals in Sediment(1)

CHEMICAL	NONCARCINOGENIC EFFECTS (mg/kg/day)		CARCINOGENIC EFFECTS (mg/kg/day)	
	DERMAL CONTACT	INGESTION	DERMAL CONTACT	INGESTION
VOLATILES				
Chloroform	4.6e-08	1.3e-09	6.6e-09	1.8e-10
SEMIVOLATILES				
Di-n-butylpthalate	6.0e-08	2.8e-06	8.5e-09	4.0e-07
Bis(2-ethylhexyl)phthalate	2.5e-08	4.0e-08	3.6e-09	5.8e-09
METALS				
Arsenic	9.1e-09	2.5e-06	1.3e-09	3.6e-07
Barium	1.9e-07	5.2e-05	2.7e-08	7.5e-06
Cadmium	2.3e-09	6.4e-07	3.3e-10	9.1e-08
Chromium (total)	2.0e-08	5.6e-06	2.9e-09	8.0e-07
Copper	1.8e-08	4.9e-06	2.5e-09	6.9e-07
Magnesium	5.5e-05	1.5e-02	7.8e-06	2.2e-03
Manganese	5.5e-07	1.5e-04	7.9e-08	2.2e-05
Nickel	2.2e-08	6.2e-06	3.2e-09	8.9e-07
Silver	2.3e-10	6.4e-08	3.3e-11	9.1e-09
Thallium	9.2e-10	2.6e-07	1.3e-10	3.7e-08
Vanadium	6.8e-11	1.9e-08	9.6e-12	2.7e-09
Zinc	1.3e-08	3.6e-06	1.8e-09	5.1e-07
Cyanide	6.1e-10	1.7e-07	8.7e-11	2.4e-08

Notes

1. a. Absorbed doses were calculated for dermal contact to sediment and intakes were calculated for ingestion of sediment.

   b. For noncarcinogenic effects, the absorbed dose (AD) or chronic daily intake (CDI) is averaged over the exposure period, while for carcinogenic effects, they are averaged over a lifetime (i.e., 70 years). Therefore, the differences between the AD or CDI for noncarcinogenic vs. carcinogenic effects are due to the different methods of time weighing which are used to estimate the value.

MWK/aao/BJC/KJD/JFK
[jlv-403-93u]
13160.40-MD

TABLE 6-17

Chronic Daily Intakes and Absorbed Doses of Chemicals in Groundwater(1)

Chemical	Noncarcinogenic Effects (mg/kg/day)		Carcinogenic Effects (mg/kg/day)	
	Dermal Contact	Ingestion	Dermal Contact	Ingestion
VOLATILES				
Acetone	2.3e-07	1.4e-04	9.6e-08	5.9e-05
Benzene	1.6e-07	9.9e-05	6.9e-08	4.2e-05
Bromodichloromethane	5.2e-09	3.2e-06	2.2e-09	1.4e-06
Bromoform	1.0e-08	6.4e-06	4.4e-09	2.7e-06
Carbon Tetrachloride	2.3e-08	1.4e-05	9.7e-09	6.0e-06
Chlorobenzene	8.5e-08	5.2e-05	3.6e-08	2.2e-05
Chloroform	1.4e-07	8.7e-05	6.1e-08	3.7e-05
Chloroethane	1.8e-06	1.1e-03	7.6e-07	4.7e-04
Chloromethane	1.1e-07	6.5e-05	4.6e-08	2.8e-05
Dibromochloromethane	6.8e-09	4.2e-06	2.9e-09	1.8e-06
1,1-Dichloroethane	3.2e-06	2.0e-03	1.4e-06	8.5e-04
1,2-Dichloroethane	6.1e-08	3.7e-05	2.6e-08	1.6e-05
1,1-Dichloroethene	1.2e-08	7.6e-06	5.3e-09	3.2e-06
1,2-Total Dichloroethene	1.1e-05	6.8e-03	4.8e-06	2.9e-03
1,2-Dichloropropane	3.4e-07	2.1e-04	1.5e-07	9.0e-05
1,3-Dichloropropene	2.0e-08	1.2e-05	8.5e-09	5.2e-06
Ethylbenzene	8.4e-08	5.2e-05	3.6e-08	2.2e-05
Methylene Chloride	1.2e-07	7.4e-05	5.1e-08	3.2e-05
Tetrachloroethene	1.1e-06	6.4e-04	4.5e-07	2.8e-04
1,1,2,2-Tetrachloroethane	1.6e-07	9.9e-05	6.9e-08	4.3e-05
Toluene	9.8e-09	6.0e-06	4.2e-09	2.6e-06
1,1,1-Trichloroethane	2.7e-07	1.7e-04	1.2e-07	7.2e-05
Trichloroethene	2.5e-06	1.5e-03	1.1e-06	6.5e-04
Vinyl chloride	6.6e-07	4.1e-04	2.8e-07	1.7e-04
Xylene	9.7e-08	6.0e-05	4.2e-08	2.6e-05
SEMIVOLATILES				
Diethylphthalate	1.1e-07	6.5e-05	4.6e-08	2.8e-05
Bis(2-ethylhexyl)Phthalate	2.1e-07	1.3e-04	9.2e-08	5.6e-05
1,2-Dichlorobenzene	6.0e-08	3.7e-05	2.6e-08	1.6e-05
1,4-Dichlorobenzene	6.9e-07	4.2e-04	2.9e-07	1.8e-04
PAHS (Noncarcinogenic)	3.9e-09	2.4e-06	1.7e-09	1.0e-06
METALS				
Arsenic	3.9e-07	2.4e-04	1.7e-07	1.0e-04
Barium	3.1e-05	1.9e-02	1.3e-05	8.1e-03
Cadmium	1.1e-07	6.5e-05	4.5e-08	2.8e-05
Chromium, (total)	3.6e-08	2.2e-05	1.5e-08	9.4e-06
Cobalt	1.5e-06	8.9e-04	6.2e-07	3.8e-04
Manganese	6.2e-05	3.8e-02	2.6e-05	1.6e-02
Nickel	1.8e-06	1.1e-03	7.5e-07	4.6e-04
Nitrates	5.4e-07	3.3e-04	2.3e-07	1.4e-04
Silver	1.0e-07	6.1e-05	4.3e-08	2.6e-05
Thallium	1.3e-07	8.0e-05	5.6e-08	3.4e-05
Vanadium	1.4e-06	8.3e-04	5.8e-07	3.6e-04
Zinc	3.0e-04	1.8e-02	1.3e-04	7.8e-02
Cyanide	4.4e-06	2.7e-03	1.9e-06	1.2e-03

Notes

1. a. Absorbed doses were calculated for dermal contact to groundwater and intakes were calculated for ingestion of groundwater.

   b. For noncarcinogenic effects, the absorbed dose (AD) or chronic daily intake (CDI) is averaged over the exposure period, while for carcinogenic effects, they are averaged over a lifetime (i.e., 70 years). Therefore, the differences between the AD or CDI for noncarcinogenic vs. carcinogenic effects are due to the different methods of time weighing which are used to estimate the value.

TABLE 6-18

Predicted Dermal Absorption Factors (ABS) For Organic Analytes from Sediment(1)

CHEMICAL	DERMAL ABS (100%= 1)
VOLATILES	
Chloroform	1.0e+00
SEMIVOLATILES	
Di-n-butylpthalate	5.9e-04
Bis(2-ethylhexyl)phthalate(2)	1.7e-02

NOTES

(1) The dermal absorption factors were estimated by predicting the proportion of sediment-bound chemical which would be in solution when the sediment is applied to the skin. The portion of the chemical which was predicted to be in solution (i.e., contained in skin surface moisture) was assumed to be completely absorbed into the body. The proportion of the chemical in solution was calculated based on the organic carbon partition coefficient (K_{OC}) and an assumed proportion or organic carbon (i.e., 1%) in sediment. The following equation was used to estimate the dermal absorption factor.

$$ABS = \frac{1}{(K_{OC} \times OC)}$$

Where: ABS = dermal absorption factor (unitless, 1 = 100%)
K_{OC} = organic carbon partition coefficient (unitless assuming the density of water and sediment is similar) (See Table 6-9)
OC = proportion of organic carbon in sediment (i.e., assume 0.01)

If a value greater than 1 (100%) is estimated, 1 was used as an estimate of the ABS.

(2) The K_{OC} value reported for this phthalate was provided by the Illinois EPA (August 3, 1990).

MWK/aao/TJD/JFK
[jlv-403-93m]
13160.40-MD

TABLE 6-19

Hazard Quotients and Cancer Risks Due to
Surface Water Exposure - Current
Land Use Conditions

CHEMICAL	HAZARD INDICES	CANCER RISKS
	DERMAL CONTACT	DERMAL CONTACT
VOLATILES		
Benzene	ND	5.8e-11
1,2-Total Dichloroethene	3.6e-07	ND
Chloroform	1.5e-07	2.0e-12
Methylene Chloride	1.2e-06	1.2e-10
1,1,1-Trichloroethane	2.1e-08	ND
Trichloroethene	ND	6.3e-12
METALS		
Barium	9.5e-05	ND
Cyanide	3.2e-06	ND
Total(all chemicals by route)	1e-04	2e-10

Legend

ND = Value not estimated because toxicity value has not yet been determined by the U.S. EPA or it is not applicable (i.e., slope factor for noncarcinogen).

MWK/kml/BJC/KJD/JFK
[jlv-403-93r]
13160.40-MD

TABLE 6-20

Hazard Quotients and Cancer Risk Due to
Sediment Exposure - Current Land Use Conditions

CHEMICAL	HAZARD INDICES		CANCER RISKS	
	DERMAL CONTACT	INGESTION	DERMAL CONTACT	INGESTION
VOLATILES				
Chloroform	9.2e-06	1.3e-07	8.0e-11	1.1e-12
SEMIVOLATILES				
Di-n-butylpthalate	1.2e-06	2.8e-05	ND	ND
Bis(2-ethylhexyl)phthalate	2.5e-06	2.0e-06	1.0e-10	8.1e-11
METALS				
Arsenic	9.1e-06	2.5e-03	2.3e-09	6.5e-07
Barium	7.5e-05	1.0e-03	ND	ND
Cadmium	2.3e-04	1.3e-03	ND	ND
Chromium (total)	6.8e-07	5.6e-06	ND	ND
Manganese	1.4e-04	1.5e-03	ND	ND
Nickel	2.2e-05	3.1e-04	ND	ND
Silver	1.5e-06	2.1e-05	ND	ND
Thallium	2.6e-04	3.7e-03	ND	ND
Vanadium	1.9e-07	2.7e-06	ND	ND
Zinc	2.6e-07	1.8e-05	ND	ND
Cyanide	4.4e-08	8.5e-06	ND	ND
TOTAL (all chemicals by route)	7.5e-04	1.0e-02	2.5e-09	6.5e-07
TOTAL (all chemicals and routes)		1e-02		6e-07

Legend

ND = Value not estimated because toxicity value has not yet been determined by the U.S. EPA.

MWK/kml/BJC/KJD/JFK
[jlv-403-93t]
13160.40-MD

TABLE 6-21

Hazard Quotients Due to
Groundwater Exposure - Potential Future Land Use Conditions

Chemical	Hazard Quotients				
	Dermal Contact	Ingestion	Inhalation(1)	Total (all routes by chemical)	% of Total Risk (all routes by chemical)
VOLATILES					
Acetone	2.8e-06	1.4e-03	1.4e-03	2.8e-03	0
Benzene	ND	ND	ND	0.0e+00	0
Bromodichloromethane	5.2e-07	1.6e-04	1.6e-04	3.2e-04	0
Bromoform	1.0e-06	3.2e-04	3.2e-04	6.4e-04	0
Carbon Tetrachloride	6.5e-05	2.0e-02	2.0e-02	4.0e-02	1
Chlorobenzene	8.5e-06	2.6e-03	2.6e-03	5.2e-03	0
Chloroethane	ND	ND	ND	0.0e+00	0
Chloroform	1.4e-05	8.7e-03	8.7e-03	1.7e-02	0
Chloromethane	ND	ND	ND	0.0e+00	0
Dibromochloromethane	6.8e-07	2.1e-04	2.1e-04	4.2e-04	0
1,1-Dichloroethane	6.5e-05	2.0e-02	2.0e-02	4.0e-02	1
1,2-Dichloroethane	ND	ND	ND	0.0e+00	0
1,1-Dichloroethene	2.7e-06	8.4e-04	8.4e-04	1.7e-03	0
1,2-Dichloroethene (total)	1.1e-03	6.8e-01	6.8e-01	1.4e+00	26
1,2-Dichloropropane	ND	ND	ND	0.0e+00	0
1,3-Dichloropropene	1.3e-04	4.1e-02	4.1e-02	8.2e-02	2
Ethylbenzene	1.7e-06	5.2e-04	5.2e-04	1.0e-03	0
Methylene Chloride	2.9e-06	1.2e-03	1.2e-03	2.5e-03	0
Phenol	1.1e-06	6.8e-04	6.8e-04	1.4e-03	0
Tetrachloroethene	2.1e-04	6.4e-02	6.4e-02	1.3e-01	2
1,1,2,2-Tetrachloroethane	ND	ND	ND	0.0e+00	0
Toluene	4.9e-08	3.0e-05	3.0e-05	6.0e-05	0
1,1,1-Trichloroethane	3.0e-06	1.9e-03	1.9e-03	3.7e-03	0
Trichloroethene	ND	ND	ND	0.0e+00	0
Vinyl chloride	ND	ND	ND	0.0e+00	0
Xylene	4.9e-08	3.0e-05	3.0e-05	6.0e-05	0
SEMIVOLATILES					
Diethylphthalate	2.7e-07	8.1e-05	ND	8.2e-05	0
Bis(2-ethylhexyl)Phthalate	2.1e-05	6.6e-03	ND	6.6e-03	0
1,2-Dichlorobenzene	1.2e-06	3.8e-04	ND	3.8e-04	0
1,4-Dichlorobenzene	ND	ND	ND	0.0e+00	0
PAHS (Noncarcinogenic)	1.9e-06	5.9e-04	ND	5.9e-04	0
METALS					
Arsenic	3.9e-04	2.4e-01	ND	2.4e-01	5
Barium	1.2e-02	3.8e-01	ND	3.9e-01	7
Cadmium	1.1e-02	1.3e-01	ND	1.4e-01	3
Chromium (total)	1.2e-06	2.2e-05	ND	2.3e-05	0
Cobalt (2)	ND	ND	ND	ND	0
Manganese	1.5e-02	3.8e-01	ND	3.9e-01	7
Nickel	1.8e-03	5.4e-02	ND	5.6e-02	1

TABLE 6-21

Chemical	Hazard Quotients				
	Dermal Contact	Ingestion	Inhalation(2)	Total (all routes by chemical)	% of Total Risk (all routes by chemical)
Nitrate + Nitrite	1.1e-04	3.3e-03	ND	3.4e-03	0
Silver	6.7e-04	2.0e-02	ND	2.1e-02	0
Thallium	3.7e-02	1.1e+00	ND	1.2e+00	22
Vanadium	3.9e-03	1.2e-01	ND	1.2e-01	2
Zinc	5.9e-03	9.1e-01	ND	9.1e-01	17
Cyanide	3.1e-04	1.3e-01	ND	1.4e-01	3
TOTAL (all chemicals by route)					
TOTAL (all chemicals and routes)	9.0e-02	4.4e+00	8.5e-01	5e+00	

Notes:

(1) Based on a study by Cothern, et al. (1986), residential exposures to VOCs through the inhalation route while bathing/showering can be equal in magnitude to exposure through the ingestion of contaminated water. Therefore, the oral estimates of risk (i.e., HI) for VOCs were used to estimate the inhalation risk due to exposure to VOCs while bathing. It was assumed the semivolatile and inorganic chemicals do not volatilize appreciably from water, and therefore, this route of exposure was not considered applicable for these classes of chemicals.

(2) Cobalt was detected in two wells, P1(84 ug/L and P4R (63 ug/L in duplicate sample only, not detected in investigative sample) during Round 1 groundwater sampling. Due to the fact that cobalt was detected in a limited number of wells and the interim oral RFD (see Appendix I) appears to be unrealistically low (i.e., below a normal daily human intake of cobalt), inclusion of cobalt in the calculation of the future groundwater estimate is not believed to be appropriate. However, if cobalt is included, it would represent 95% of the noncancer risk, resulting in a Hazard Index = 100.

Legend

ND = Value not estimated because a U.S. EPA verified toxicity value has not yet been determined.

TABLE 6-22

Cancer Risks Due to
Groundwater Exposure - Potential Future Land Use Conditions

Chemical	Cancer Risks				
	Dermal Contact	Ingestion	Inhalation(1)	Total (all routes by chemical)	% of Total Risk (all routes by chemical)
VOLATILES					
Acetone	ND	ND	ND	0.0e+00	0
Benzene	4.0e-09	1.2e-06	1.2e-06	2.5e-06	0
Bromodichloromethane	5.8e-10	1.8e-07	1.8e-07	3.5e-07	0
Bromoform	7.0e-11	2.2e-08	2.2e-08	4.3e-08	0
Carbon Tetrachloride	2.5e-09	7.7e-07	7.7e-07	1.6e-06	0
Chlorobenzene	ND	ND	ND	0.0e+00	0
Chloroethane	ND	ND	ND	0.0e+00	0
Chloroform	3.7e-10	2.3e-07	2.3e-07	4.5e-07	0
Chloromethane	1.2e-09	3.6e-07	3.6e-07	7.3e-07	0
Dibromochloromethane	4.9e-10	1.5e-07	1.5e-07	3.0e-07	0
1,1-Dichloroethane	ND	ND	ND	0.0e+00	0
1,2-Dichloroethane	2.4e-09	1.5e-06	1.5e-06	2.9e-06	0
1,1-Dichloroethene	6.3e-09	1.9e-06	1.9e-06	3.9e-06	0
1,2-Dichloroethene (total)	ND	ND	ND	0.0e+00	0
1,2-Dichloropropane	2.0e-08	6.1e-06	6.1e-06	1.2e-05	1
1,3-Dichloropropene	3.1e-09	9.4e-07	9.4e-07	1.9e-06	0
Ethylbenzene	ND	ND	ND	0.0e+00	0
Methylene Chloride	5.5e-10	2.4e-07	2.4e-07	4.7e-07	0
Phenol	ND	ND	ND	0.0e+00	0
Tetrachloroethene	4.6e-08	1.4e-05	1.4e-05	2.8e-05	3
1,1,2,2-Tetrachloroethane	1.4e-08	8.5e-06	8.5e-06	1.7e-05	2
Toluene	ND	ND	ND	0.0e+00	0
1,1,1-Trichloroethane	ND	ND	ND	0.0e+00	0
Trichloroethene	1.2e-08	7.2e-06	7.2e-06	1.4e-05	1
Vinyl chloride	1.3e-06	4.0e-04	4.0e-04	8.0e-04	74
Xylene	ND	ND	ND	0.0e+00	0
SEMIVOLATILES					
Diethylphthalate	ND	ND	ND	0.0e+00	0
Bis(2-ethylhexyl)Phthalate	2.6e-09	7.9e-07	ND	7.9e-07	0
1,2-Dichlorobenzene	ND	ND	ND	0.0e+00	0
1,4-Dichlorobenzene	1.4e-08	4.3e-06	ND	8.7e-06	1
PAHS (Noncarcinogenic)	ND	ND	ND	0.0e+00	0

TABLE 6-22

Chemical	Cancer Risks				
	Dermal Contact	Ingestion	Inhalation(1)	Total (all routes by chemical)	% of Total Risk (all routes by chemical)
METALS					
Arsenic	3.0e-07	1.8e-04	ND	1.8e-04	17
Barium	ND	ND	ND	0.0e+00	0
Cadmium	ND	ND	ND	0.0e+00	0
Chromium (total)	ND	ND	ND	0.0e+00	0
Cobalt	ND	ND	ND	0.0e+00	0
Manganese	ND	ND	ND	0.0e+00	0
Nickel	ND	ND	ND	0.0e+00	0
Nitrates	ND	ND	ND	0.0e+00	0
Silver	ND	ND	ND	0.0e+00	0
Thallium	ND	ND	ND	0.0e+00	0
Vanadium	ND	ND	ND	0.0e+00	0
Zinc	ND	ND	ND	0.0e+00	0
Cyanide	ND	ND	ND	0.0e+00	0
TOTAL (all chemicals by route)	1.7e-06	6.3-04	4.5e-04		
TOTAL (all chemicals and routes)			1.1e-03		

Note

(1) Based on a study by Cothern, et al. (1986), residential exposures to VOCs through the inhalation route while bathing/showering can be equal in magnitude to exposure through the ingestion of contaminated water. Therefore, the oral estimates of risk (i.e., cancer risks) for VOCs were used to estimate the inhalation risk due to exposure to VOCs while bathing. It was assumed the semivolatile and inorganic chemicals do not volatilize appreciably from water, and therefore, this route of exposure was not considered applicable for these classes of chemicals.

Legend

ND = Value not estimated because toxicity value has not yet been determined by the U.S. EPA.

Environmental Evaluation

The objectives of this component of the Baseline Risk Assessment were to characterize the natural habitats which may be influenced by the Site and to appraise the actual or potential adverse effects contaminants have had on these habitats.

Killbuck Creek and nearby wetlands were assumed to be the most sensitive ecological habitats near the landfill. The Creek is rated as a "Class B Stream- Highly Valued Aquatic Resource".

Fish were considered the most susceptible group of aquatic species to chemical exposure in Killbuck Creek. Effects on fish are not expected based on the chemical concentrations in surface water in comparison to Ambient Water Quality Criteria. Therefore, since this sensitive group of organisms appears to be safe from health effects, other aquatic ecosystem effects are not anticipated.

Health risks to the terrestrial environment could not be compared to applicable criteria, because floodplain sediment and surface soil samples were not analyzed as part of the RI. Based on visual observations, signs of impacts on the terrestrial ecosystem were not observed (e.g., stressed vegetation). Also, because of the nature of the contamination at the WRL Site (i.e., groundwater), impacts on the terrestrial ecosystem are not expected.

SECTION 7
SUMMARY AND CONCLUSIONS

7.1 Site Characteristics

The WRL Site, also known as Pagel's Pit, is an active solid waste landfill licensed by the State of Illinois, and has been operating since 1972. The WRL Site has a liner and leachate collection system, and a landfill gas collection system. The central and eastern two-thirds of the landfill have received an intermediate 2-foot thick vegetated cover (final cover will be placed at the time of landfill closure).

Land use around the site is a mix of industrial, agricultural, commercial, and rural residential. East (upgradient) of the WRL Site is the former Acme Solvent Reclaiming Inc. (Acme Solvent) Site, which was used for the disposal of solvent still bottom sludges, nonrecoverable solvents, paints and oils into unlined lagoons as well as the burial of drums.

Wastes accepted at the WRL Site are composed primarily of municipal refuse and sewage treatment plant sludge. A limited amount of Illinois special non-municipal wastes were disposed of at the Site. The WRL leachate is characterized by its high inorganic component, particularly the chloride ion content. The WRL leachate generally contained low levels of aromatic VOCs such as benzene, toluene, ethylbenzene, and xylenes at higher concentrations than chlorinated compounds such as PCE and TCE.

The thickness of the unconsolidated materials range from 8 feet in the bedrock uplands to the east of the WRL Site to greater than 70 feet at the western boundary of the WRL Site. The unconsolidated materials are dominantly sand and gravel or fine to coarse sand. The bedrock underlying the unconsolidated materials is composed of fractured dolomite. The bedrock surface elevation is highly variable due to paleoerosional features. A bedrock valley begins to form under the WRL Site and deepens to the west.

Groundwater flows from the bedrock uplands east of the WRL Site, west towards the Killbuck Creek Valley. Groundwater at depth flows west beneath Killbuck Creek, but shallow groundwater discharges to the Creek. East of the WRL Site and below approximately the eastern quarter, the water table occurs in the fractured dolomite bedrock. West of the eastern one quarter of the WRL Site, the water table occurs in the unconsolidated materials.

Groundwater flow within the dolomite bedrock is believed to be largely controlled by fractures. The presence of a high permeability zone in the vicinity of the elevation interval 690 ft to 710 ft MSL is inferred from geologic data, geophysical data, pumping test results, permeability test results, water level observations, and groundwater quality data. The effect of this high permeability zone on contaminant transport is to confine the VOCs to discrete zones at the upgradient end and disperse the VOCs widely in the dolomite aquifer at the downgradient end of the zone. This hypothetical pattern is consistent with the observed VOC distributions in the dolomite aquifer.

The WRL leachate has a high inorganic component (chlorides) that serve as a conservative "tracer" of leachate migration from the landfill. The presence of elevated chloride concentrations in the groundwater is found in three areas around the WRL landfill. These areas are:

- The northwest quadrant of the WRL Site defined by wells B15R, MW106, P1, P4R, and G116A
- In the vicinity of wells G110 and G114
- In the vicinity of well G115

In the northwest quadrant of the WRL Site, the WRL leachate groundwater plume is only present in the shallow groundwater near well nest B15/B15R/B15P, but has not affected the deeper groundwater as indicated by the low levels of chlorides observed in the deeper well B15P. The WRL leachate groundwater plume has migrated west of the WRL Site and has impacted both the shallow and deeper groundwater zones east of Killbuck Creek as indicated by the observed chloride concentrations in samples from the well nest P1/MW106. The WRL leachate groundwater plume had migrated beneath Killbuck Creek and impacted the deeper groundwater but not the shallow groundwater as indicated by the observed chloride results from well G116A. Rounds 3 and 4 groundwater samples from well nest G116/G116A indicate the WRL leachate plume impacts have been reduced, since the chloride concentrations have decreased.

The second groundwater area exhibiting elevated chlorides is the vicinity of wells G110 and G114. The observed chloride concentrations in samples from well G110 have continually increased from Round 1 to Round 4 indicating increasing influence of WRL leachate on the groundwater in the vicinity over this period. In addition, the observed chloride concentrations in samples from nearby well G114 were clearly elevated in Rounds 3 and 4, indicating the impacts of WRL leachate expanded to include groundwater in the vicinity of well G114 as well as G110. The impacts of WRL leachate in this area are limited to areas upgradient of well nest B13/P6 since samples from these wells have not exhibited clearly elevated levels of chlorides, but rather have shown little change in observed chloride levels.

The third groundwater area exhibiting elevated chlorides is in the vicinity of well G115. The observed chloride concentrations in the groundwater increased between Rounds 2 and 3 to levels clearly indicating the presence of WRL leachate. This trend was confirmed in Round 4 groundwater analysis, indicating an increasing presence of WRL leachate at the southwestern margin of the landfill.

The WRL leachate groundwater plume inorganic component tends to contain elevated chlorides, sodium, potassium, magnesium, manganese, and iron. Other constituents sometimes associated with the WRL leachate plume include: total phenolics, cyanide, arsenic, barium, cobalt, copper, lead, nickel, silver, vanadium, and zinc. These constituents were not detected or were present at only low levels in samples from well G116A, indicating these constituents are attenuated in the aquifer.

Volatile organic compounds (VOCs) and semi-volatile organic compounds (SVOCs) are present both upgradient and at the WRL Site and both inside and outside the WRL plume defined by elevated chlorides. The groundwater upgradient of the WRL Site has been impacted by VOCs with the highest concentrations being observed in samples from well B4. The VOCs observed in the RI samples can be divided into three general groups: VOCs associated with well B4, VOCs associated with wells upgradient of the WRL Site near Lindenwood Road, and VOCs present only in the WRL leachate plume.

The detection of VOCs from the east to west direction (i.e., direction of groundwater) based upon the RI samples is as follows. The VOCs detected at well B4 (hydraulically upgradient of the WRL Site) include chlorinated ethenes, chlorinated ethanes, 1,1-dichloropropane, chloromethane, and BETX (benzene, ethylbenzene, toluene and xylenes). The VOCs associated with wells upgradient of the WRL Site near Lindenwood Road but not found at well B4 based upon RI samples include chlorobenzene, trans-1,3-dichloropropene, and dibromochloromethane. The VOCs detected only in locations downgradient of the WRL Site include low levels of carbon tetrachloride, bromoform, chloroform, bromodichloromethane, and acetone.

No SVOCs were detected in RI samples from well B4. The general group of SVOCs first associated with wells upgradient of the WRL near Lindenwood Road include: 1,2-dichlorobenzene, 1,4-dichlorobenzene and bis (2-ethylhexl) phthalate. The SVOCs detected only in sample locations downgradient of the WRL Site include: 1,3-dichlorobenzene, acenaphthene, and dibenzofuran. No pesticides or PCBs were detected in any of the RI groundwater samples.

Based upon upstream-downstream trends in the RI data and published information, the WRL Site has not impacted the surface water or sediments of Killbuck Creek. Based upon the RI data, the ambient air quality does not appear to be affected by the WRL.

VOCs are present both inside and outside of the limits of the WRL leachate groundwater plume as defined by elevated chlorides. This indicates that the presence of VOCs outside of the WRL leachate plume are not due to WRL leachate. Potential other sources of VOCs to the groundwater include releases from the upgradient Acme Solvent Site, migration of landfill gas, and a localized source such as effluent from the household septic systems in the immediate vicinity.

The disposal practices at the Acme Solvent Site have resulted in the release of VOCs to the groundwater as documented by the detection of VOCs in groundwater samples from upgradient wells. The Acme Solvent Site is located hydraulically upgradient from the WRL Site and so releases to the groundwater at the Acme Solvent Site are expected to be transported in a downgradient direction towards the WRL Site. The extent of the presence of VOCs has not been adequately defined by the existing monitoring well network downgradient of the Acme Solvent Site. It is recommended that an additional study be conducted to document the continuity of the VOC plume between the two sites.

7.2 Contaminant Fate and Transport

The fate and transport of contaminant compounds identified at, or adjacent to, the WRL site are dependent upon the chemical and physical characteristics of the contaminants, characteristics of the contaminated media, source of contamination, climate, and potential migration pathways. Groundwater transport is believed to be the major pathway by which contaminants are transported in the vicinity of the WRL Site, regardless of the source. Surface water, sediment, subsurface gas and air are other possible routes by which contaminants may be transported, but are of lesser importance based upon current WRL Site conditions.

Adsorption of VOCs in the unsaturated zone beneath the landfill or in the aquifer to the sand and gravel materials is not expected to play a major role in the downgradient transport of VOCs. Attenuation of metals and SVOCs in the saturated or unsaturated zones is expected to be greater than that of the VOCs because of their lower mobility. Attenuation due to dilution and biodegradation for organic compounds can be expected to reduce concentrations of contaminants in the downgradient direction.

There is no indication that Killbuck Creek has been impacted by groundwater discharge to the Creek. If shallow contaminated groundwater were discharged to the Creek, it is expected that significant dilution would occur. VOCs, if present, would be volatilized during downstream transport. Dissolved metals and SVOCs may be adsorbed onto suspended sediment, transported downstream and deposited.

Landfill gas extraction is the primary soil vapor transport pathway, and the gas is currently burned as a fuel for the sludge drying plant. Low level air emissions through the landfill would be quickly attenuated during downwind transport.

Other potential mechanisms of contaminant transport (i.e., soil erosion, fugitive air emissions, and surface water movement) were evaluated, but are not considered to be substantial mechanisms of chemical transport under current and probable future use conditions at the WRL.

7.3 Baseline Risk Assessment (BRA)

The objective of the Baseline Risk Assessment (BRA) is to characterize the extent of contamination and the potentially exposed human and ecological populations sufficiently to determine what risks need to be prevented. The BRA is composed of a Human Health Evaluation and an Environmental Evaluation.

Human Health Evaluation

A Human Health Evaluation was conducted to estimate the risk people may incur as a result of exposure to chemicals present at the WRL Site. This assessment was conducted without regard to the source(s) of the Site contamination. A risk assessment was made based on both current and potential future Site conditions.

Under the current use scenario, surface water and sediment in Killbuck Creek appear to pose the most likely point of chemical exposure to individuals (children) living in the area of the WRL Site. Other media exposures were considered in the BRA under current conditions (i.e., air, water, food); however, these media are not expected to contribute substantially to human chemical exposure.

Based on current Site conditions, it was assumed children will play in Killbuck Creek and may be exposed to sediment by incidental ingestion and contact, and to surface water via direct contact. Assuming these exposure conditions, noncarcinogenic health effects are not expected (i.e., $HI < 1$) and cancer risks are low (i.e., $< 1x10^{-6}$).

In the future, there is the potential for exposure conditions to change if land use practices change. Based upon possible future land use conditions, persons may use the groundwater near the WRL Site as a source of drinking water. Under future use considerations, exposures of humans to other media (i.e., air, food) were considered, but were not expected to contribute substantially to human chemical exposure.

Under a hypothetical future use scenario, it was assumed that residents would be exposed via ingestion, as well as dermal contact and inhalation, to the contaminants in groundwater. Unlike under current Site conditions, noncarcinogenic health effects may be of concern and cancer risks are substantially greater than the U.S. EPA's risk range ($1x10^{-4}$ to $1x10^{-6}$), assuming residents were exposed daily to the chemicals of potential concern in groundwater for 30 years.

The primary noncarcinogenic health risks under the future use scenario (84%) were associated with 1,2-dichloroethene (26%), arsenic (5%), barium (7%), manganese (7%), thallium (22%), and zinc (17%) resulting in a noncarcinogenic risk of HI=5. Interactive effects are possible for 1,2-dichloroethene and zinc; therefore, the combined exposure to these chemicals may pose a low potential for noncarcinogenic blood effects (e.g., anemia) to occur in humans.

On the other hand, the majority (91%) of the cancer risks were due to exposure to arsenic (17%) and vinyl chloride (74%). Both chemicals are known human carcinogens.

Currently, the WRL Site does not appear to present a public health concern. However, potential future exposure to groundwater at the WRL Site appears to present a potential public health concern based upon estimates of noncancer and cancer risk.

Although groundwater exposure at the WRL Site is above U.S. EPA Risk Goals, the health risk at the WRL Site is similar to anthropogenic levels of risk associated with the off-site upgradient groundwater contamination. Therefore, although future potential exposure to groundwater at and downgradient of the WRL Site appears to be a potential public health concern, anthropogenic background appears to contribute to this risk.

SECTION 8
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SDMS ADMINISTRATIVE RECORD IMAGERY INSERT FORM

SITE NAME	PAGELS PIT
DOC ID #	70446 / Page 274
DESCRIPTION OF ITEM(S)	MAPS
REASON WHY UNSCANNABLE	____ ILLEGIBLE or __X__ FORMAT OVERSIZED
DATE OF ITEM(S)	04-02-1991
NO. OF ITEMS	2
PHASE	__X__ Remedial _____ Removal _____ Deletion Docket Volume __7__ of __9__ _____ Original ____ Update # ___
O.U.	
FRC	Box # __1__ Folder # __7__
COMMENTS	